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Report of the Working Group on Fish Ecology (WGFE)

2–7 April 2004
ICES, Copenhagen

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EXECUTIVE SUMMARY

This report details the work undertaken for the second meeting of the ICES Working Group on Fish Ecology (WGFE). The work areas undertaken addressed the following issues: Ecological Quality Objectives (EcoQOs) for fish communities (Section 2), threatened and declining fish species and abundance-range size relationships (Section 3), prey composition, food rations and gastric evacuation of North Sea gadoids (Section 4), fish habitat (Section 5), the relative catchability of fishes in different survey gears (Section 6) and preliminary studies examining the status of North Sea fish communities (Section 7).

Ecological Quality Objectives (EcoQOs) for fish communities are required by OSPAR, and analyses on various size-based metrics have been undertaken by WGFE. It was originally suggested that size-based metrics would be primarily responsive to the effects of fishing activity, although current studies have indicated that environmental factors may also affect such indices. Further studies are required to develop EcoQOs for fish communities, and it is suggested that a suite of indicators, each addressing specific concerns or issues, and examined on the relevant spatial and temporal scale, should be developed. Additionally, studies to examine the relative importance of fishing and environmental variation should be encouraged.

Methods of identifying threatened and declining fish species were reviewed, and a critique of the Texel-Faial criteria, as developed by OSPAR, given. It is suggested that these criteria need some refinement, primarily in terms of developing a standardised approach to their application. Preliminary analyses of inter-specific and intraspecific abundance-occupancy relationships were undertaken, and the theoretical background to this aspect of macroecology summarised. It was considered that this area of research had potential applications to the conservation and management of fishes and fishery resources, and that further studies should be continued.

Revised estimates of the prey composition, gastric evacuation and food rations of North Sea gadoids have been made. The new estimates of total food ration differed from the old ones used at present by ICES to acquire predation mortality figures for the MSVPA prey fish populations. It is recommended that the consequences of using the new ration estimates in relevant runs of MSVPA/FOR, as compared to the results obtained from the old ration estimates, are examined, particularly with a view to re-evaluating predation mortalities of the MSVPA prey fish populations.

Various aspects of fish habitat were described, and summarised information provided for various commercial fish species. Improved knowledge of fish habitat, including the identification of any areas that are of critical importance to threatened and declining fish species, is required. Studies on fish habitat are also required to better understand abundance-occupancy relationships.

Factors that can affect the relative catchability of fishes in different survey gears have been reviewed and two case-study analyses provided. As expected, different gears have very different catchabilities for various fish species and their size composition, which has consequences for analyses of fish “communities” and associated metrics. Improved knowledge of the catchability of species and their size classes are required if better estimates of biomass are to be made. Additionally, such studies may also allow some degree of integration of relative abundance data from various survey gears for the broadscale mapping of fish distributions.

Various aspects of the fish communities of the North Sea were summarised from the available literature, and a case study of the types of analyses that can be undertaken for national and international surveys presented. Such regional analyses of fish assemblages may play an important role in the regional assessment of ecosystems.

1 INTRODUCTION

1.1 Terms of reference

The Working Group on Fish Ecology [WGFE] (Chair: J. Ellis, UK) will meet at ICES Headquarters from 2–7 April 2004 to:

- a) develop EcoQOs relating to fish communities and associated reference levels [OSPAR 2004/1] by:
 - i) reviewing the theoretical basis of size-structured indicators,
 - ii) conducting exploratory analyses of trophic level and size spectra,
 - iii) exploring the utility and application of EcoQOs over a range of spatial scales;
- b) identify threatened and declining fish species by:
 - i) reviewing existing methods of identifying rare, threatened and declining fish species, including an evaluation of the Texel-Faial criteria,
 - ii) examining abundance-range size relationships in selected fish species,
 - iii) based on i) and ii) developing a set of criteria that can be used to prioritise species in the OSPAR area that may require more detailed assessments/status reports in the future;
- c) complete studies on food rations, prey composition and gastric evacuation rates of gadoids in the North Sea;
- d) review current knowledge of habitat requirements of commercial, rare and threatened fish species, including diadromous species with particular emphasis on the distribution of critical habitats;
- e) start analyses of relative catchabilities of the more common fish species in different survey gears;
- f) start preparations to summarise status and changes in fish species distribution and fish community composition and interactions in the North Sea for the period 2000–2004, for input to the Regional Ecosystem Study Group for the North Sea in 2006.

WGFE will report by 30 April 2004 for the attention of the Living Resources, the Resource Management, and the Diadromous Fish Committees, as well as ACE.

1.2 Participants

The following scientists attended the Working Group meeting.

Full contact details are given in Annex I.

Ole Thomas Albert	Norway
Niels Gerner Andersen	Denmark
Julia Blanchard	UK
Maria de Fatima Borges	Portugal
Nick Dulvy	UK
Jim Ellis	UK
Ann-Britt Florin	Sweden
Ronald Fricke	Germany
Helen Fraser	Scotland
Henk Heessen	The Netherlands
Simon Greenstreet	Scotland
Bart Maertens	Belgium

The following working group members contributed by correspondence, and submitted valuable background documents, data or text.

Niels Daan	The Netherlands
Daniel Duplisea	Canada
Siegfried Ehrich	Germany
Dave Kulka	Canada

1.3 Background

The Working Group on Fish Ecology first met in 2003. The rationale behind the formation of the group was to support ICES on issues of fish community metrics and for providing advice on threatened marine fishes. OSPAR and HELCOM have requested advice in these areas in the recent past that ICES was unable to respond to.

Until 2002, fish community issues were considered by WGECO. The demands on WGECO were heavy and the establishment of WGFE enabled a more focussed consideration of community issues. This forum enables ICES to address issues on rarer, non-commercial fish species, and provides background so that ICES can now provide advice in this area in relation to biodiversity and nature conservation issues.

2 DEVELOPING ECOQOS RELATING FISH COMMUNITIES AND ASSOCIATED REFERENCE LEVELS

2.1 Introduction

In 2003, WGFE felt that it was not yet either appropriate to implement average weight, average maximum length or the proportion of large fish in the community as part of an EcoQO, nor to define a global North Sea reference level for management (ICES, 2003). As an incentive to further explore the appropriateness of size-based and other fish community indicators the following term of reference was established for the 2004 WGFE meeting.

ToR a: develop EcoQOs relating fish communities and associated reference levels [OSPAR 2004/1] by:

- i) Reviewing the theoretical basis of size structured indicators
- ii) Conducting exploratory analyses of trophic level and size spectra
- iii) Exploring the utility and application of EcoQOs over a range of spatial scales

There is currently a large amount of research on the development of fish community and ecosystem indicators for the ecosystem approach to fisheries management (EAFM). Work carried out by the Scientific Committee on Oceanic Research (SCOR) and Intergovernmental Oceanographic Committee (IOC) Working Group 119 “Quantitative Ecosystem Indicators for Fisheries Management” was presented at an international symposium held during 31 March–3 April, along with presentations by many international researchers on this topic. A small number of contributions presented at WGFE were simultaneously submitted to the SCOR/IOC symposium. In this ToR, the WGFE focussed on a relatively narrow range of indicators that at present are well understood, with a focus on size-based indicators and to a limited extent trophodynamic and diversity indicators. The topics presented at the SCOR/IOC international symposium comprised a much larger scope and will therefore be helpful in further considerations of EcoQOs and their evaluation and performance. The themes presented at the symposium were: (1) Environmental indicators; (2) Diversity and species indicators; (3) Size-based indicators; (4) Trophodynamic indicators; (5) Spatial indicators; (6) Evaluating indicators; (7) Integrated indicators; (8) Frameworks for sustainable development; (9) Implementation schemes; and (10) World implementation. Papers associated with this symposium will be published in the ICES Journal of Marine Science and the contributions therein are likely to tie in closely and complement the work of WGFE under this ToR. For further information on the SCOR/IOC symposium see the following website: <http://www.ecosystemindicators.org/>.

2.2 Theoretical basis of size-structured indicators

2.2.1 Introduction

Size-based approaches appear to have a suitable theoretical foundation and practical utility. Harvesting tends to be size selective such that larger individuals and species are preferentially removed by fishers and these tend to be at higher trophic levels and have higher commercial value (Jennings *et al.*, 2002; Pauly *et al.*, 1998; Pinnegar *et al.*, 2002; Polunin and Pinnegar, 2002). Larger-bodied species also tend to be more intrinsically vulnerable to exploitation because they mature later, grow slower and consequently exhibit lower maximum population growth rates and weaker density dependence (Denney *et al.*, 2002; Frisk *et al.*, 2001; Jennings *et al.*, 1998; Myers *et al.*, 1999; Myers *et al.*, 1997; Reynolds *et al.*, 2001). In addition both metabolism and production scales with body size such that larger bodied species and individuals have higher metabolic rates and lower production (Peters, 1983).

Size spectra have been suggested as a useful method of detecting the effects of exploitation upon fish communities (Pope and Knights, 1982; Pope *et al.* 1988; Murawski and Idoine 1992; Bianchi *et al.* 2000; Zwanenburg 2000). Size spectra in their simplest form can be described by the slope and intercept of a regression of the right-hand descending limb of the community size frequency distribution. It has been inferred from multispecies modelling that size-selective harvesting of larger bodied individuals and species should result in a decline in the slope of size spectra as fishing effects become pronounced (Gislason and Rice, 1998). Declining slopes and increasing intercepts have been observed over time in a variety of shelf systems (Bianchi *et al.*, 2000; Duplisea *et al.*, 1997; Jennings *et al.*, 2002; Pope *et al.*, 1988; Rice and Gislason, 1996). In the macroecological literature size spectra are referred to as abundance-body size distributions and there is a large body of theory associated with these, providing an opportunity for links between the two approaches (Jennings and Mackinson, 2003).

2.2.2 Size spectra theory and methodology

Size spectra are multispecies metrics of the abundance or biomass of organisms plotted as a function of their body size. Duplisea and Castonguay (submitted) described the theory and practical utility of different methods. Size spectra data are usually statistically fitted with a regression, the parameters of which are used to characterise the system under scrutiny. Size spectra have been used in fisheries to show that exploitation steepens a straight-line slope over time (Pope *et al.* 1988; Bianchi *et al.* 2000), which indicates a systemic decrease in the abundance of large fish that are directly removed by fisheries and can also correspond to an indirect effect of fishing through an increase in small fish due to release from predation.

Though patterns in size spectra can indicate trends in exploited fish communities, there are several confusing aspects of size spectra; most of the confusion arises out of methodological and statistical treatment of data, as well as the applicability of certain theory (Duplisea and Castonguay, submitted). Three types of spectra have been explored:

- 1) \log_2 body weight classes v. \log_2 biomass (Boudreau *et al.* 1991; Duplisea and Kerr 1995),
- 2) normalized biomass spectra where the biomass of each body weight class is divided by the width of the body weight class (Platt and Denman 1977, 1978), and
- 3) 5 cm length class bins – frequency of \log_2 numbers (Pope *et al.* 1988).

Linear models have been fit to methods 2 and 3 whereas a quadratic model has been used to describe the first type of spectra. The choice of which statistical model to fit to size spectra is determined by both shape of data and theory. Initial attempts to generalize size-spectra of slope 0 to all organisms in the sea created an incentive to try and fit straight lines and examine deviations from the 0 slope. (Kerr 1974) later pointed out that periodic deviations with size exist about this straight line and proposed that these arose out modes in predator-prey size ratios and affected standing stocks of particular size classes. Platt and Denman (1978) furthered the models of Sheldon *et al.* (1972) and Kerr (1974) and developed a normalised biomass spectrum. Furthermore, Platt and Denman (1978) theoretically derived a straight line slope prediction for the normalised biomass spectrum ($b = -1.22$) based on allometric parameterisation of respiration, production and that predators consume prey in the adjacent smaller size class. Han and Straškraba (1998) discuss the implications of several of the above size spectrum approaches for both theoretical and empirical analyses.

An alternative analytical model used in size spectra describes a quadratic superimposed by a series of identical sub-quadratics corresponding to trophic groups (phytoplankton, zooplankton, fish) (Thiebaut and Dickie 1993). This model was an extension of the original Platt and Denman (1978) model where periodic solutions were modelled as quadratic. Hence, though straight lines are simpler to fit and use fewer degrees of freedom than parabolas, both empirical and theoretical work support the fitting of parabolas as well as straight lines, depending on data type (Duplisea and Castonguay, submitted). The key issue is whether all size classes have been sampled equally within the size range. For example, the biomass of smaller size classes tends to be underestimated and requires the inclusion of small pelagic fishes and benthic infauna and epifauna (Jennings and Dulvy, In press).

Purely statistical fitting methods without *a priori* assumptions of spectrum shape are now being employed to describe observed size spectrum patterns. Pareto distributions were shown to represent normalised size spectrum slope but were considered more favourable given that fewer assumptions were made in their fitting (Vidondo *et al.* 1997). The method of non-parametric kernel density estimators reveals positions of peaks and troughs in size spectra (Havlicek and Carpenter 2001). This method makes no assumption about spectrum shape yet assumes that real ecological information exists in the observed patterns of peaks and troughs and attempts to focus on these aspects of size spectra rather than overall trends.

The theoretical work has enabled size spectra to be used in a predictive sense in addition to the static descriptions that can be obtained from statistical fits of empirical size spectra. The size spectrum theory relies on the flow of biomass

from the smallest to the largest sized organisms through size dependent processes. Some of the theoretical models consider discrete trophic levels and the processes that have been considered include growth, production, respiration, predation and reproduction (Kerr 1974, Borgman 1982, 1983, Thiebaut and Dickie 1993). Platt and Denman 1978 and Silvert and Platt (1978, 1980) developed a continuous flow model that avoided difficulty associated with defining trophic levels. They developed a continuous time-dependent nonlinear model of the size spectrum governed by predation and including growth and mortality processes. Their results showed that biomass spectra are linear when a fixed predator prey size ratio is assumed. Benoit and Rochet (2004) recently developed a new continuous model allowing for the assumption of fixed predator prey ratio to be relaxed where predation is distributed across sizes. Their results indicated that linearity is a property of this model and that fishing steepens the slope of size spectra above a size threshold, and increases the curvature of the entire spectrum. They compare their results to other simulation studies; one developed with an MSVPA structure (Gislason and Rice 1998) and the other an individual based model (Shin and Cury, in press). This area is rapidly developing as the quest for reference levels, responses of fish communities and the effects of fishing on properties of biomass spectra continues.

2.2.3 Size spectra versus abundance-body mass relationships

Theory has developed in parallel in the terrestrial macroecological literature. Here size spectra are known as abundance – body size distributions. In the terrestrial literature there has been a tendency to explore the abundance –body size distributions of taxonomically defined groups, e.g., birds or mammals (Damuth 1981; Blackburn and Lawton 1995; Brown 1995; Cotgreave 1993; Gaston and Blackburn 2000). In aquatic systems subsets of communities or even whole communities have been studied, e.g., marine benthos, freshwater lakes, plankton and demersal fish (e.g., Schwinghamer 1981; Sprules *et al.* 1983; Boudreau *et al.*, 1991). The fundamental difference between the two approaches is that in terrestrial systems the unit of analysis has been individual species and in aquatic ecosystems the unit of analysis has been particle size or body size. This recognizes the fact that aquatic systems are highly size-structured - nearly all species grow continuously throughout their lives, exhibiting indeterminate growth (Charnov and Berrigan, 1991) and thus begin life in the small size classes and grow in size by up to five orders of magnitude (Cushing 1975). By contrast most of the taxa studied in terrestrial systems stop growing in size at maturity (determinate growth).

The relationship of the numerical density (numbers per km²) versus body mass (g) relationship for herbivorous mammals was linear on a logarithmic scale and the slope was –0.75 (Damuth 1981). From this it was concluded that for animals feeding on a common energy source there is a power relationship between numerical density (N) and body mass (M):

$$N \propto M^{-0.75}$$

It is widely known that metabolic rate (R) scales with body mass (M):

$$R \propto M^{0.75}$$

Damuth (1981) combined these two relationships to find that energy use (E) per unit area scales as:

$$E \propto M^{-0.75} \times M^{0.75} \propto M^0$$

This leads to the conclusion that species and individuals in an assemblage using a common energy source use approximately similar amounts of energy. In reality while larger species may have access to more energy the cost of their higher individual metabolism results in lower abundance compared to smaller species or individuals. This has been called the ‘energetic equivalence’ rule (Nee *et al.* 1991). Many communities and taxonomic groupings exhibit slopes departing from –0.75, this is because such ‘samples’ violate a key assumption – they do not share a common energy source (Brown and Gillooly 2003; Jennings and Mackinson 2003).

In reality the energy available decreases with each increasing trophic level due to the inefficient transfer efficiency (TE) between trophic levels. Much energy is not converted to growth, but is lost due to excretion and metabolism, hence only between 5–30% of the energy in one trophic level is available to consumers in the trophic level above (Pauly and Christensen 1995; Ware, 2000). In heavily size structured aquatic systems trophic levels are not categorical, but continuous. It is possible for an individual consuming differing amount of a number of prey species, each existing at a number of trophic levels, can exhibit a fractional trophic level. This pattern can be expressed as the slope of the relationship between body mass and trophic level, which can be measured using nitrogen stable isotope ratio (Fry and Quinones 1994; Jennings *et al.* 2001). The slope of the body mass – trophic level relationship is called the predator prey mass ratio ($PPMR$).

Hence in size structured communities the amount of energy available for a given body mass is:

$$E \propto MTE / PPMR$$

Note that both TE and PPMR are \log_{10} transformed. Thus the numerical density at a given body mass is:

$$N \propto MTE / PPMR \times M^{-0.75}$$

And since biomass density scales with body mass (M) as

$$B \propto M^{0.25},$$

and therefore the scaling between biomass density and body mass in a size structured ecosystem is:

$$B \propto M^{TE / PPMR} \times M^{0.25}.$$

Hence size structured ecosystems should display numerical size spectra slopes > -0.75 for the relationship between numbers ($\log_{10} n \text{ m}^{-2}$) and body mass ($\log_{10} \text{ g}$) and < 0.25 for the relationship between biomass ($\log_{10} \text{ g m}^{-2}$) and body mass ($\log_{10} \text{ g}$). In an analysis of the relatively unexploited part of the North Sea fish community using all animals between 2–256 g wet mass, Jennings and Mackinson (2003) observed scalings of:

$$N \propto M^{-1.2}$$

$$B \propto M^{0.2}.$$

Predator-prey mass ratio was measured using nitrogen stable isotopes assuming a mean fractionation of 3.4 parts per 1000 as;

$$PPMR = 10^{(3.4/\text{slope})},$$

where the slope was derived from the relationship between body mass and $\delta^{15} \text{ N}$. The PPMR used was 106:1 and the transfer efficiency was assumed to be 0.1. Based on these estimates the predicted slopes should be:

$$E \propto M^{\log_{10}(0.1) / \log_{10}(109)} = -0.49$$

$$N \propto M^{-0.49} \times M^{-0.75} = -1.24$$

$$B \propto M^{-0.49} \times M^{0.25} = -0.24$$

The observed slopes (see Figure 2.2.3.1) were not significantly different from the values predicted above using the modified energetic equivalence model. The predicted scaling of numbers and biomass are relatively insensitive to the range of transfer efficiencies (Figure 2.2.3.2.).

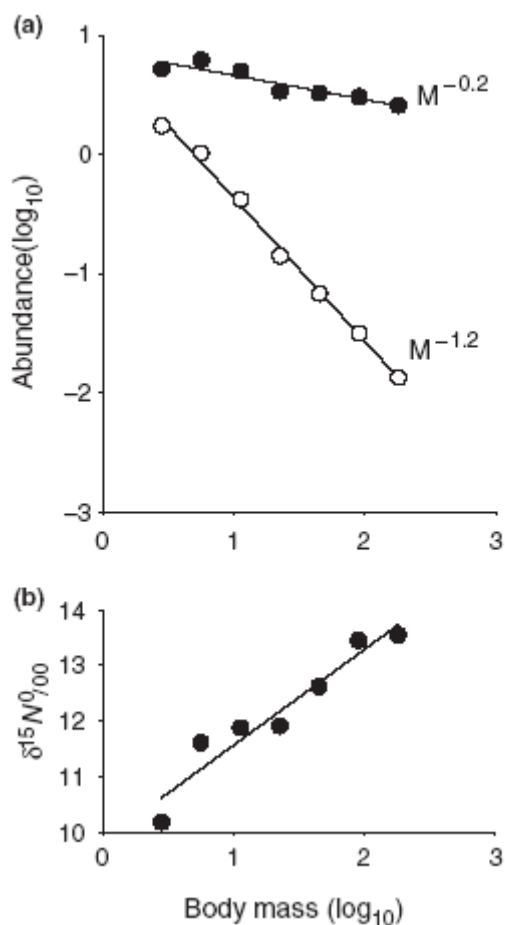


Figure 2.2.3.1 (a) The relationship between biomass abundance (filled circles) or numerical abundance (open circles) and body mass in the unexploited part of a North Sea food web. (b) The relationship between $\delta^{15}N$ and body mass.

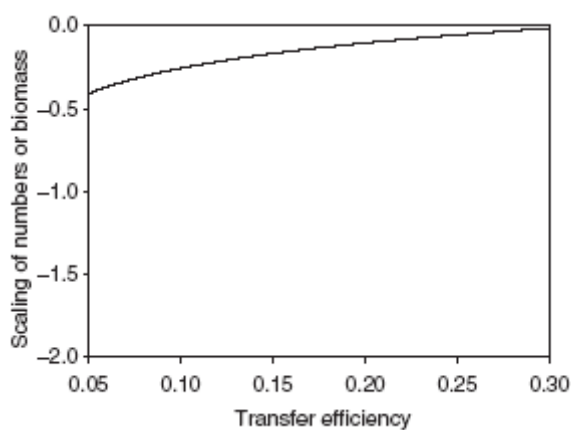


Figure 2.2.3.2. Predicted scaling of numbers as a function of transfer efficiency.

For typical ranges of PPMR (102–103:1) and TE (0.1–0.2) slopes of the unexploited size spectrum ($\log_{10} B$ versus $\log_{10} M$) are predicted to range from approximately 0 to -0.2. This is far shallower than the slopes of size spectra in most fished ecosystems. The approach described in Jennings and Mackinson (2003) provided a method for setting unexploited reference points for slope of size spectrum and mean body mass. Jennings and Blanchard (In Press) applied the approach in the North Sea and compared the contemporary fish community with the same fish community in the absence of fishing. PPMR was calculated from the slope of the relationship between trophic level and M for fishes in the survey trawl catches and a range of TE values were used (Ware, 2000). For the calculated scaling of E and M and assuming a TE of 0.125, the predicted slope of the size spectrum was 0.1. There were clear differences between the contemporary and unexploited size spectrum slopes and mean individual body mass in the contemporary fish community was 144 g, 38% of that as predicted for the unexploited community (Jennings and Blanchard, In Press). These analyses were based on fish sampled from the entire community and therefore applying the same methods to routine survey data would require knowing the gear catchability of size compositions to be correctly compared with unexploited reference points. Further work on reconciling the theories associated with macroecological approaches and size spectra is warranted, especially in considering the development of reference levels.

2.3 Case studies of exploratory analyses of trophic level and size spectra

2.3.1 Introduction

Exploratory analyses of size spectra were undertaken for data available on the Celtic Sea, the Portuguese Shelf and the Barents Sea. These are reported in the following sections as case studies.

2.3.2 Case Study of the Celtic Sea (ICES Divisions VIII-f-j)

2.3.2.1 Introduction

Although the slope of size spectra steepen over time in a manner that is interpreted as being consistent with exploitation-mediated changes in community structure, there has been little consideration of how sampling gear and ecological factors also influence these and other size based metrics. Gear type, and seasonal and spatial differences have been shown to produce significant differences in size-based metrics across given time series (Daan *et al.* in press; Trenckel *et al.* submitted). Recent studies found decreases in larger fish were accompanied by increases in smaller fish, which could be interpreted as the result of predatory or competitive release (Dulvy *et al.* in press, Daan *et al.* in press, ICES 2003). In addition to this, the relative importance of environmental effects, and the interaction between these and exploitation are difficult to quantify, and both factors may lead to changes in size structure.

For size to be a meaningful indicator of community structure the metrics derived should ideally be able to discriminate between differing impacts or forcing factors acting upon community structure, such as exploitation, climatic events or regimes, pollution and the background variation. Here we explore size metrics of the Celtic Sea fish assemblage both over time and in combination with measures of fishing effort and climate variability (winter sea-surface temperature)

Three size based metrics were used in our analyses: average weight of individuals, average maximum size (L_{\max}), and slopes of size spectra (Bianchi *et al.*, 2000; Duplisea *et al.*, 1997; Jennings *et al.*, 1999b; Rice and Gislason, 1996; Rochet and Trenkel, 2003). Temporal and spatial patterns in the resulting size-based metrics were examined for the Celtic Sea fish community and compared to patterns in available data for sea temperature and exploitation indices. These analyses are reported in Blanchard *et al.* (2003).

2.3.2.2 Methods

Survey data were collected from the annual CEFAS Celtic Sea groundfish surveys (RV Cirolana) (Warnes and Jones, 1995). Only locations that were sampled with a standard survey trawl throughout the time series and data for years where there was adequate spatial coverage were used in our analyses. The resultant time series spanned 1987–2003.

The temperature series the three-month winter mean (mean value Dec-March) for each year and these data were obtained from the following website: <http://www.cdc.noaa.gov/coads/>. Gridded spatial data (1° by 0.5°) for sea surface and near sea bottom temperatures and total international landings of the main commercial species in the Celtic Sea were obtained from ICES. A multispecies fishing mortality index was calculated as the biomass-weighted mean of estimated fishing mortality (F) for the major commercial species in the Celtic Sea according to an approach described by Daan *et al.* (in press). Spatial fisheries surveillance data were standardised for the amount of sightings effort within an ICES rectangle per flight and were obtained from UK Department of Environment, Fisheries and Rural Affairs (Defra).

Average weight was calculated as

$$\overline{W} = \sum W / N$$

where W is the body mass of an individual and N is the total number of individuals (Zwanenberg, 2000).

Average community L_{\max} was calculated as

$$\overline{L_{\max}} = \sum (L_{\max j} N_j) / N$$

where $L_{\max j}$ is the maximum length obtained by species j, and N is the number of individuals of species j.

Linear regression was used to derive slopes and intercepts of the normalised log 2 biomass (y-axis) versus the midpoints of log2 body mass classes (x-axis) for each year. The original data were standardised catch numbers by length categories. For the purposes of this analysis, the gear used in this survey (Portuguese high-headline trawl) was assumed to have a consistent catchability for all fish larger than 4 g; therefore only weight classes above this threshold were used to fit the linear size spectra. Lengths were transformed to individual weights by using species-specific length-weight regression coefficients (Bedford *et al.*, 1986; Dorel, 1986 and Coull *et al.*, 1989) and for species that did not have sufficiently local weight-length relationships a standardised equation was used ($W=0.01 \cdot L^3$). Size spectra were standardised to remove the correlation between slope and intercept by centring the independent values (Rochet and Trenkel, 2003).

Positive or negative trends in the size-based metrics and log-abundance over time were evaluated by comparing the rank test correlation for each time series (Spearman's correlation coefficient, rho). Differences between metrics were evaluated on the comparison of test coefficients and significance level. Localised smoothing (LOESS) of time series was used to describe trends over time.

Spatial analyses of size-based metrics were restricted to biomass spectra. Survey data were pooled over time and linear fits to log 2 normalised biomass versus log 2 body mass were made for each station that was sampled consistently over the 1987–2003 period. Spatial maps of temperature and standardised sightings of fishing vessels (proxy for fishing effort) were produced using kriging in Surfer 7.0.

2.3.2.3 Results

Temporal analyses

There were negative trends over time in average weight (rho =−0.57, p=0.0221), average L_{\max} (rho =−0.45, p=0.0736), and slopes of size spectra (rho =−0.69, p=0.005) (Figure 2.3.2.1). The observed pattern in average weight and size spectra slopes exhibited a ‘dip’ during 1993–1996. Metrics based on considering only demersal fish exhibited a decline in average weight (rho=−0.54176, p=0.029), L_{\max} (rho=−0.59, p=0.019), and slope of size spectra over time (rho=−0.53, p=0.03).

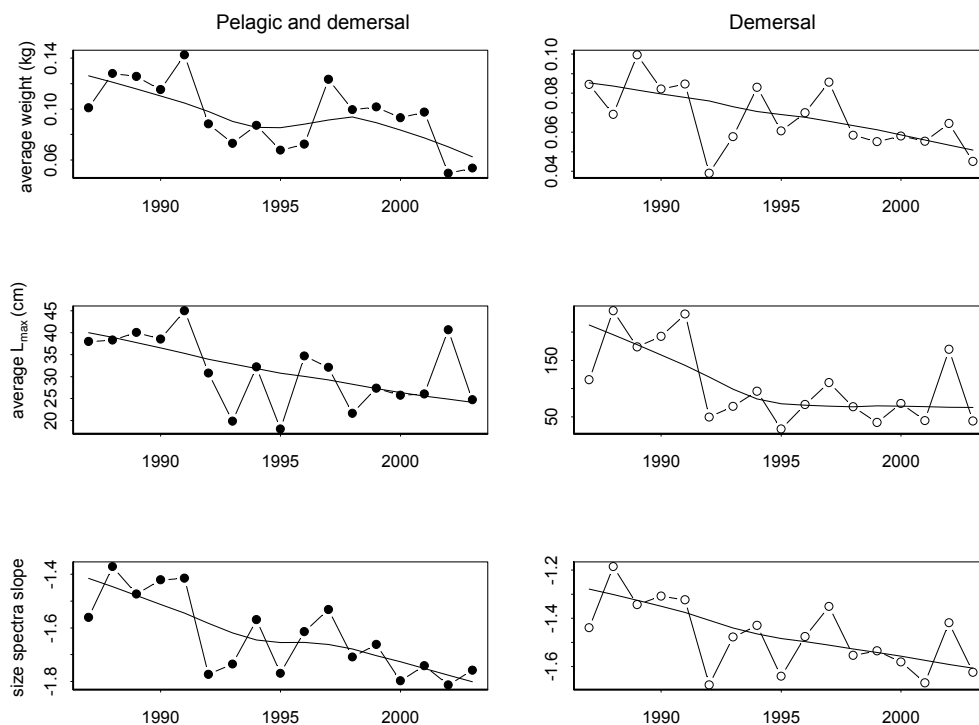


Figure 2.3.2.1. Time series plots of average weight (top row), average L_{\max} (middle row) and slopes (bottom row) of size spectra for pelagic and demersal species combined (first column) and demersal species separately (second column) in the Celtic Sea. Line fits are LOESS local smoother.

There was a significant increase over time in winter sea surface temperature for the Celtic Sea over the study period ($\rho=0.502$, $p=0.045$). There were striking similarities in the temporal pattern of sea surface temperature and the pattern in size structure of the fish community over time (Figures 2.3.2.1 and 2.3.2.2).

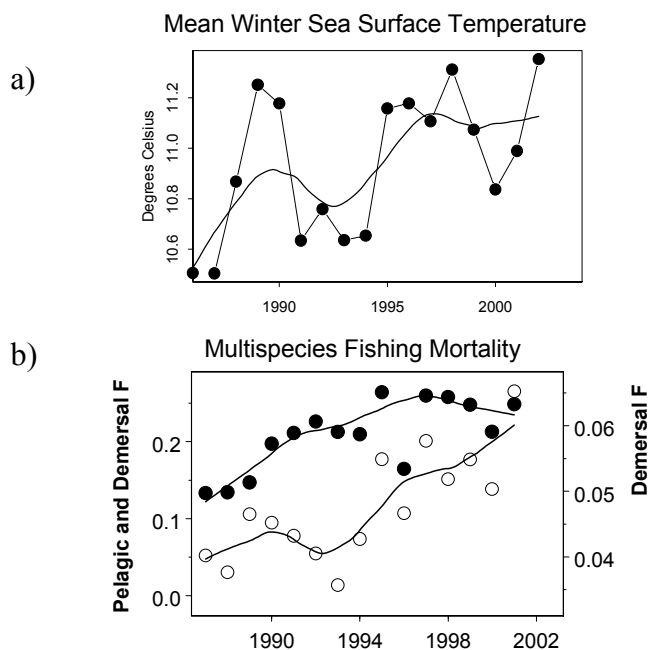


Figure 2.3.2.2. Time series of (a) winter sea surface temperature over the entire Celtic Sea (lines show LOESS locally smoothed trend), and (b) multispecies F (weighted by biomass) over time for all species combined (solid circles) and demersal species (open circles) grouped separately.

The multispecies F index for the entire Celtic Sea fish survey data increased over time ($\rho = 0.74$, $p = 0.006$), although there was a drop in 1996. The multispecies F for demersal species appears to have increased more over time ($\rho = 0.76$, $p = 0.0043$), even though values of F are much lower for this group compared to the weighted average for all fish species (Figure 2.3.2.2). Community L_{\max} , and the slopes of the size spectra were both significantly negatively correlated ($\rho = -0.73$, $p = 0.006$; $\rho = -0.58$, $p = 0.03$) with multispecies F.

Average weight and average L_{\max} and size spectra slopes were not significantly correlated with winter sea surface temperature when the biological data were not lagged. The interannual pattern of winter sea surface temperature series appeared similar to that of the size-based metrics when a 2-year lag was assumed. However, the long-term trends were opposing due to the patterns at the beginning and end of the time series. Addition of a 2 year lagged winter sea surface temperature into a multiple regression model, following either year or multispecies F as the first predictor variable did not result in significant p-values for the coefficient associated with sea surface temperature. Removal of the longterm trend in the lagged sea surface temperature time series and including the short term or inter-annual deviations from the long-term temperature also did not explain any significant additional variation in the size-based metrics. A 2-year running mean of the size spectra slopes for the Celtic Sea fish community was, however, negatively correlated with a 2-year running mean of sea temperature (2 years previous) ($\rho = -0.55$, $p = 0.03$). The danger with this approach is that we are imposing somewhat arbitrary lags to remove background variation on both variables and we are contrasting this with the hypothesized lag effects of temperature on average weight. Also, it is well known that the addition of a single year of observations can degrade these relationships and correcting for autocorrelation results in loss of significance of the correlation, especially when the series is short (<20 yr), as is the present time series (Drinkwater and Myers, 1987).

Relative biomass of log 2 weight classes revealed an increase in small fish over time that has been accompanied by a reduction in large fish (Figure 2.3.2.3). All size classes below log 2 body mass of 6.5 showed increases over time and the first three of these were significant ($p < 0.05$). Decreasing trends in the biomass of large size classes (above 6.5) were evident although only the 6.5 and 12.5 log 2 body mass classes declined significantly over time ($p < 0.05$).

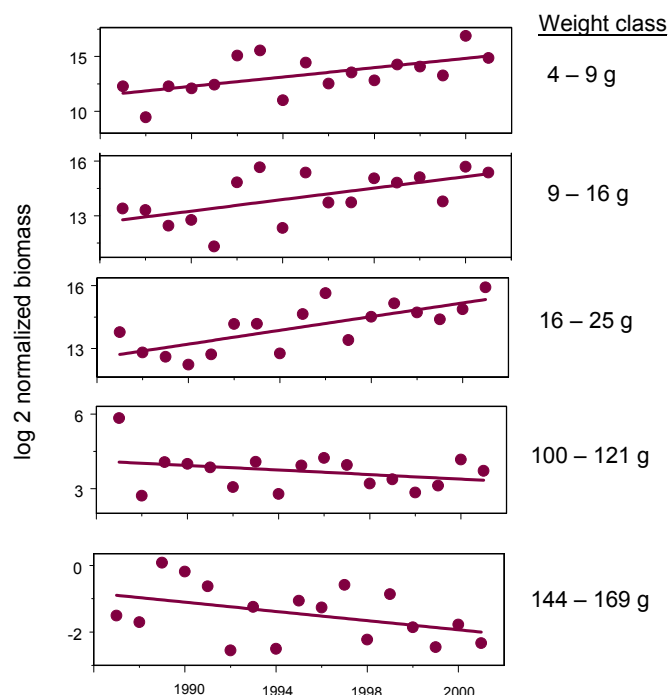


Figure 2.3.2.3. Relative biomass of selected weight classes for the entire Celtic Sea fish community over time.

Pooling the time series data and fitting linear biomass spectra models to each prime station sampled consistently over the time period revealed that the steepest slopes were located at stations sampled in the western part of the Celtic Sea (along the shelf edge) and in the eastern part of the Celtic Sea. The majority of stations sampled in the central areas of the Celtic Sea had relatively less steep slopes. Within the spatial surveillance limits of the fishing vessel sightings data, areas that were demonstrated to have highest fishing intensity were consistent with those areas that exhibited steepest size spectra slopes spatially. Overall, the area where relatively low fishing intensity was visually apparent (the central area of the Celtic Sea) corresponded to the areas where size spectra slopes were shallowest (Figure 2.3.2.4). Interestingly, spatial areas with steepest size spectra slopes also corresponded to areas along the shelf edge where thermal stratification and adjacent high productivity are known to occur (Figure 2.3.2.4). Stations that have undergone the greatest change (steepening) in slopes over time were located in the central part of the Celtic Sea (Figure 2.3.2.5).

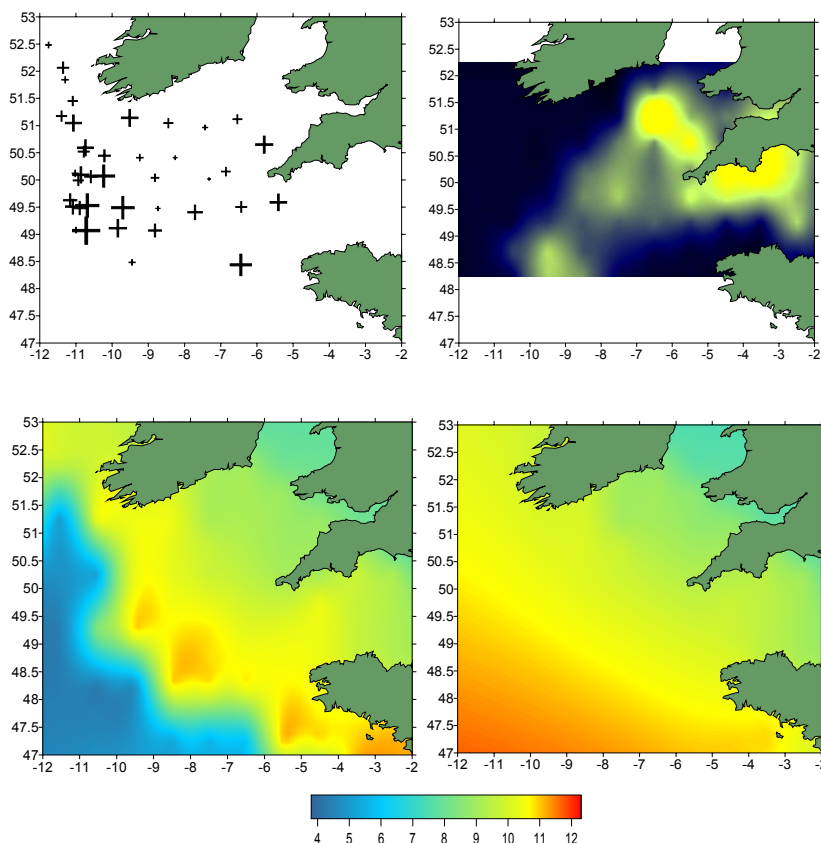


Figure 2.3.2.4. Slopes of biomass spectra mapped by station for all stations sampled consistently in the survey over 1987–2003 (top left). Size of crosses is proportional to slope; a larger cross indicates a steeper (more negative) size spectrum slope. Standardised sightings of UK fishing vessels in the Celtic Sea (top right, pooled data for 1985–2002) Surveillance area does not extend past bright blue contour. Mean near sea bottom temperature for March (bottom left). Mean sea surface temperature for March (bottom right).

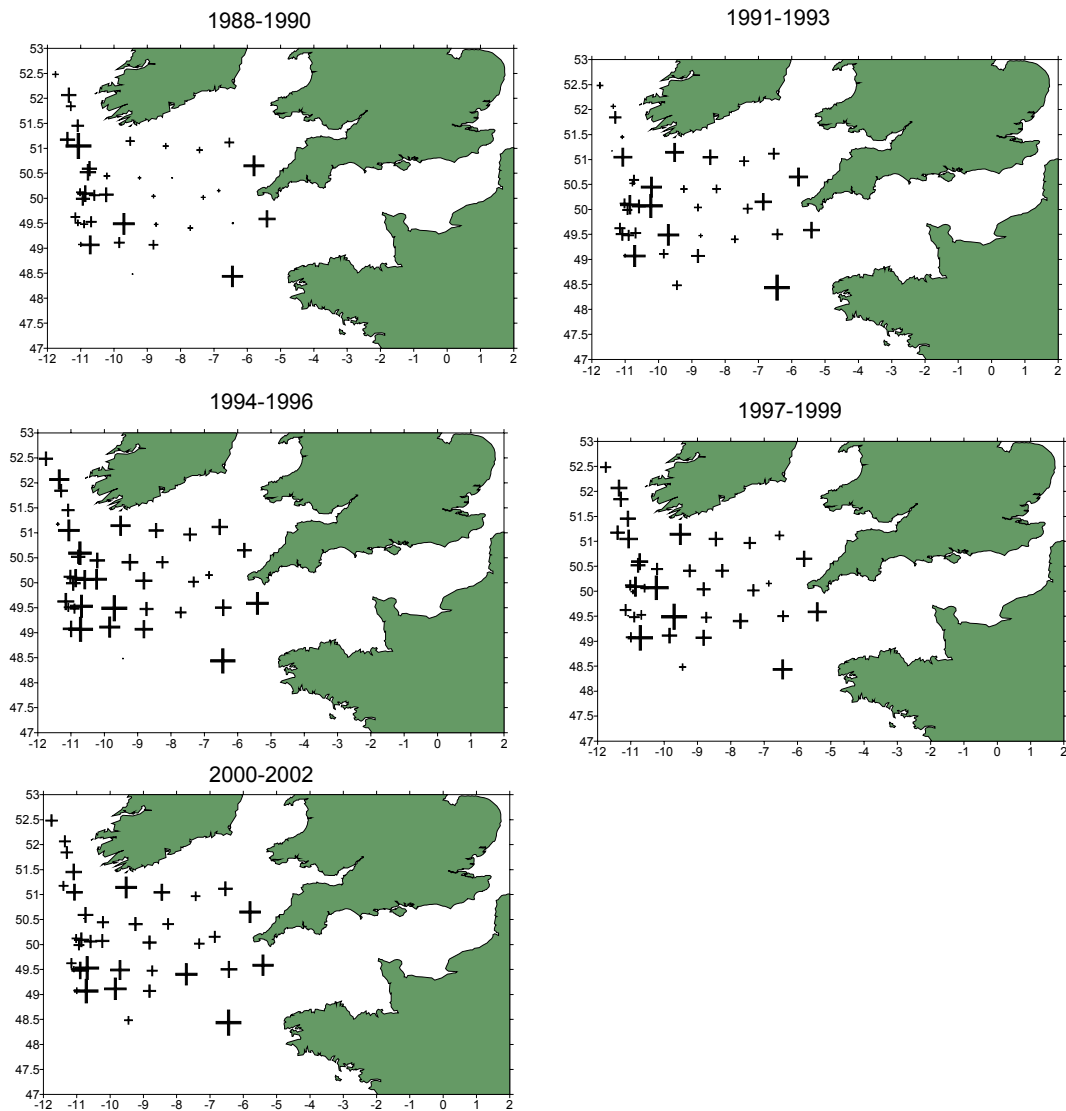


Figure 2.3.2.5. Spatio-temporal patterns in biomass spectra slopes for stations sampled consistently over time in the Celtic Sea, data pooled by 3 year periods.

2.3.2.4 Discussion

Size-based indicators appear to be useful in that they are easily understood, the effects of fishing on size based metrics are well known, and (for size spectra) the underlying ecological theory is well-developed. However, the ability for the indicator to reflect changes driven by a particular forcing factor (in this case exploitation) and the sensitivity of the indicator to other factors (i.e., environmental processes) needs to be clearly understood before a full evaluation of size-based metrics is achieved. Although size-based metrics have been shown to be a useful means of describing the changes of fish communities due to exploitation (Bianchi *et al.*, 2000; Duplisea *et al.*, 1997; Jennings *et al.*, 2002; Pope *et al.*, 1988; Rice and Gislason, 1996), they are not immune to the influence of other factors (Daan *et al.* unpublished; Trenckel *et al.* submitted).

Our analyses have shown that patterns in size-based metrics are evident both spatially and temporally. These appear to be influenced by a combination of exploitation and environmentally driven 'signals' operating at different temporal and spatial scales. There were observed declines in average weight, average L_{\max} and slopes of size spectra over time when all species were considered in the Celtic Sea. These declines have resulted from the combination of a reduction in larger

fish and an increase in smaller fish over time. The former may be the direct effect of increased fishing intensity over time. The latter could be an indirect effect of the removal of the larger size classes possibly manifested by either decreased predation or competitive 'release'.

Significant direct relationships were evident between our size-based metrics (L_{\max} and size spectra slopes) and our measures of fishing intensity multispecies F . Spatially, there also appeared to be consistencies between size spectra slopes and fishing intensity within the UK surveillance area. However, it is possible that the standardised sightings data we used to represent fishing effort do not represent the true fishing intensity of the area given that these only reflected data from one country and other countries target different species (Trenckel *et al.* submitted). International data for describing the total fishing effort at such a spatial scale is required for future analyses of the effects of fishing. In addition, higher resolution spatio-temporal physical data than presently available for the Celtic Sea would also allow for more detailed analyses of the interactions by mapping changes in the fish community, environmental and fisheries effort over space and time.

Increases in small sizes accompanied by the reduction of large individuals have been shown in the North Sea (Daan *et al.*, in press) and for Fijian coral reefs (Dulvy *et al.*, in press). A possible combination of fishing and environmental effects acting on small sizes further complicates processes, since changes in primary production can be caused by environmental factors and anthropogenic influences other than fishing (e.g., eutrophication).

We exercised caution in attempting to correlate the size-based indices with environmental variables, as such relationships are known to be unstable and often degrade over time (Drinkwater and Myers, 1987). First of all, deciding on the most appropriate temporal and spatial scale for the environmental index is difficult, especially when we are concerned with aggregate indicators of a community that integrate a variety of processes over a variety of temporal and spatial scales.

Irrespective of the species, temperature is likely to have different scales of lag effects on different sizes and parts of the community sampled since turnover time is exponentially related to body size (Banse and Mosher, 1980). Attempting to predict the effect of one single environmentally driven factor (at one particular lag-time) on aggregate measures of the fish community therefore presents obvious difficulties. Also, the statistical power associated with shorter time series of size-based indicators is problematic for analyses (Jennings and Dulvy, in press).

The slopes of size spectra differed spatially, with steeper negative slopes occurring along the shelf edge and in the eastern region of the Celtic Sea. Areas in the eastern region, such as the Bristol Channel, are known to be important nursery areas for juvenile fish and are also productive at particular times of the year. Whether this is a result of higher productivity due to the nearby presence of fronts (Simpson, 1981; Le Fèvre, 1986) requires further work with more detailed physical data.

When a change is observed in the slopes of the size spectra and other size-based metrics, this may be attributed to the effects of fishing and/or the environment. Since the location of trawl survey stations is fixed in space, changes that may be attributable to the environment may represent a change in the community present at the station (due to the effects of the environment on distribution) or an effect on the dynamics of the populations that comprise the community.

Distributional changes in marine organisms in response to climate are well known in the Celtic Sea (e.g., Beaugrand *et al.*, 2000, 2002). Southward *et al.* (1988) reported shifts in the distributions of pilchard *Sardina pilchardus* (a warmer water species) and herring *Clupea harengus* (a colder water species), and such shifts have apparently occurred over the past 400 years, corresponding with long-term temperature fluctuations.

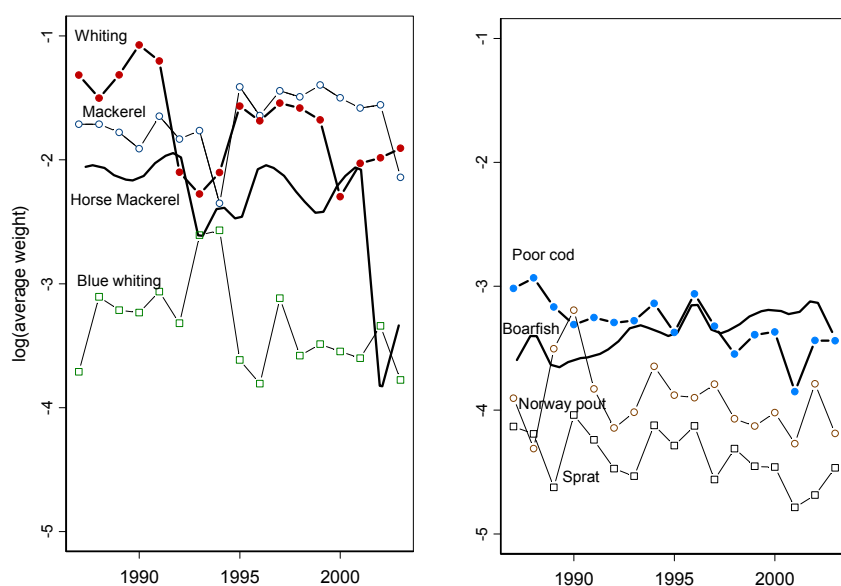


Figure 2.3.2.6. Patterns in log (average weight) of the most abundant species (representing 95% of ranked total fish species abundance) in the Celtic Sea over time (1987 – 2003). Species names are indicated from top to bottom.

Boarfish *Capros aper* have become increasingly abundant in the Celtic Sea in recent warmer years (Pinnegar *et al.* 2002) and this trend has been accompanied by a steady increase in the average weight of boarfish over time (Figure 2.3.2.6). The increase in abundance has been noted by other authors throughout the Northeast Atlantic (Farina *et al.* 1997, Abad and Giráldez, 1990) and could be the result of changing environment, increased available ecological niche, or potentially other factors that influence favourable conditions for reproductive success (Pinnegar *et al.* 2002, Trenckel *et al.* submitted). Alternatively, the increase in blue whiting *Micromesistius poutassou* during the early 1990s may represent changes in migration patterns of that particular species during colder years.

Other responses to climate may be driven by environmental conditions that can influence growth, survival and food availability during early life history. These include temperature effects of growth rate. Temperature effects may carry through the life history and if average growth rate changes across the community there are implications for age/size at maturity of fish in the community, reduced generation time and faster turnover time of individuals. Temperature, the timing of algal blooms (O'Brien *et al.* 2000, Planque and Frédou, 1999, Platt *et al.*, 2003) and the North Atlantic Oscillation signal (Attrill and Power, 2002; Dippner, 1997) have been correlated with fluctuations in marine fish populations. These processes are thought to influence the variability in the production of fish stocks that comprise the community. Theoretical and/or simulation modelling of hypothesized processes influenced by temperature (such as bioenergetics and system metabolism) combined with exploitation effects could help to shed light on the sensitivity of the metrics when both changes are occurring simultaneously.

The influence of large-scale environmental factors related to climate change could be influencing fish assemblages over a much larger area considered here (i.e., the entire Northeast Atlantic Shelf). Understanding how these factors interact to influence the structure of communities and ecosystems at different spatial scales would appear to be an important prerequisite for the effective development of size-based ecosystem and community metrics.

2.3.3 Case study of L_{max} and trophic level in Portuguese continental waters (ICES Division IXa)

2.3.3.1 Introduction

Fishing affects fish communities through selective removal of target species, through the by catch of non-target species and through habitat modification, resulting in changes in overall biomass, species composition and size structure. The extent of the response depends on the life-history characteristics of the individual species, trophic interactions among species, and on the type of changes generated in the physical habitat. Fishing will affect different species or individuals

of the same species to a greater or lesser extent and this depends of species life-history characteristics. Large species are in general associated with high trophic levels (Pauly *et al.* 2001), slow growing species, low fecundity, older age of first maturity.

Following earlier studies (ICES 2002, ICES 2003), the utility of indicators based on L_{\max} , abundance indices, diversity metrics and trophic level were explored for data collected for Portuguese waters (Hill and Borges, working paper, Annex II).

2.3.3.2 Methods and Results

Temporal patterns in L_{\max} for different fish assemblages

Fish assemblages were categorised based on L_{\max} , as opposed to the more commonly used species composition. This is taken to be the maximum observed size in all surveys. This approach assumes that what is structuring the assemblages reflects historic pressure on the constituents, including fishing pressure. Assemblages were identified using k-means clustering on the number of individuals of each 5cm length group. Hartigan's rule was used to determine the ideal number of clusters (Hartigan 1975). Due to the abundance of boarfish and snipefish, these species were omitted from data analyses.

Boxplots were then plotted for depth (10m groups) and latitude (1° groups) to determine which factors explained the assemblages best. It can be seen that depth was the most important factor (Figure 2.3.3.1a), except between assemblages 3 and 4, which was explained by latitude (Figure 2.3.3.1b).

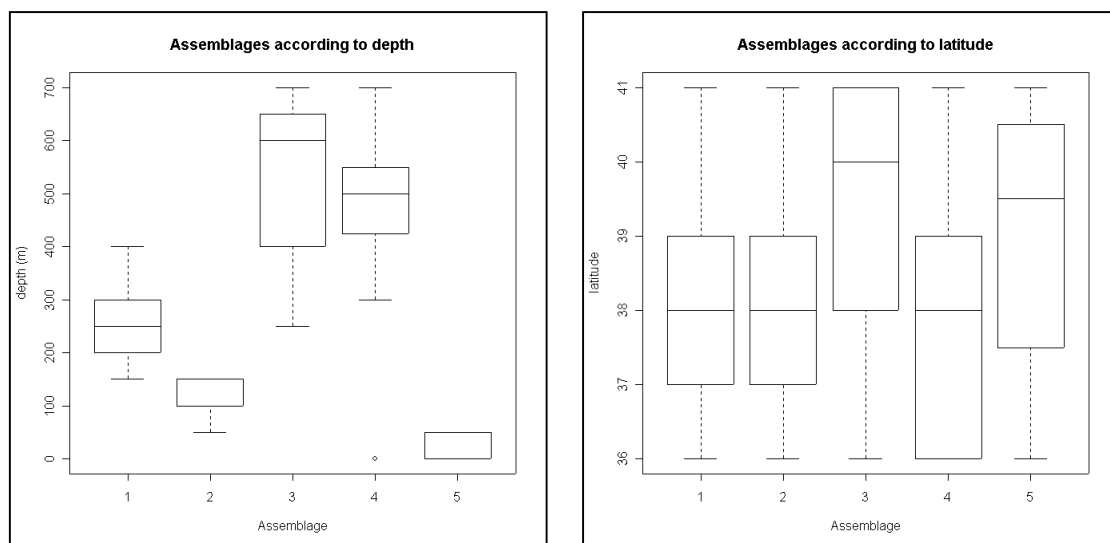


Figure 2.3.3.1. Boxplots of assemblages according to (a) depth and (b) latitude.

Three metrics were tested for assemblages 1, 2, 4 and 5. Assemblage 3 was only sampled in a few surveys, and so no metrics were applied to this assemblage. The metrics were:

- Average L_{\max} over time – the weighted average of logged L_{\max} was calculated using the number of individuals of each size caught, for each year and for each assemblage (Figure 2.3.3.2).
- Change in abundance of small, medium and large fish over time – the natural log (ln) of numbers per hour was plotted for fish <30cm, 30–50cm and >50cm for each assemblage (Figure 2.3.3.3)
- Trends in diversity indices of L_{\max} over time – L_{\max} richness (S), Simpson's Index of diversity (1–D), the Shannon Wiener index (H) and Evenness (E) were calculated for the average L_{\max} each year for each assemblage (Figure 2.3.3.4).

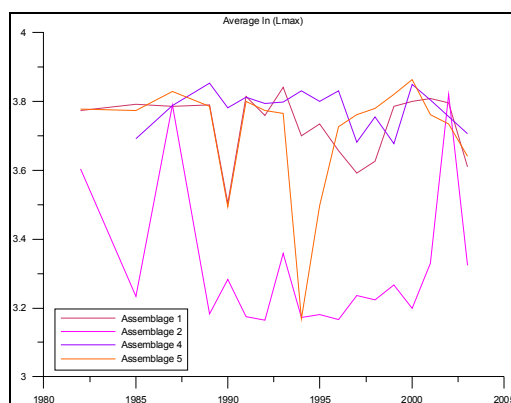


Figure 2.3.3.2. Average L_{\max} over time.

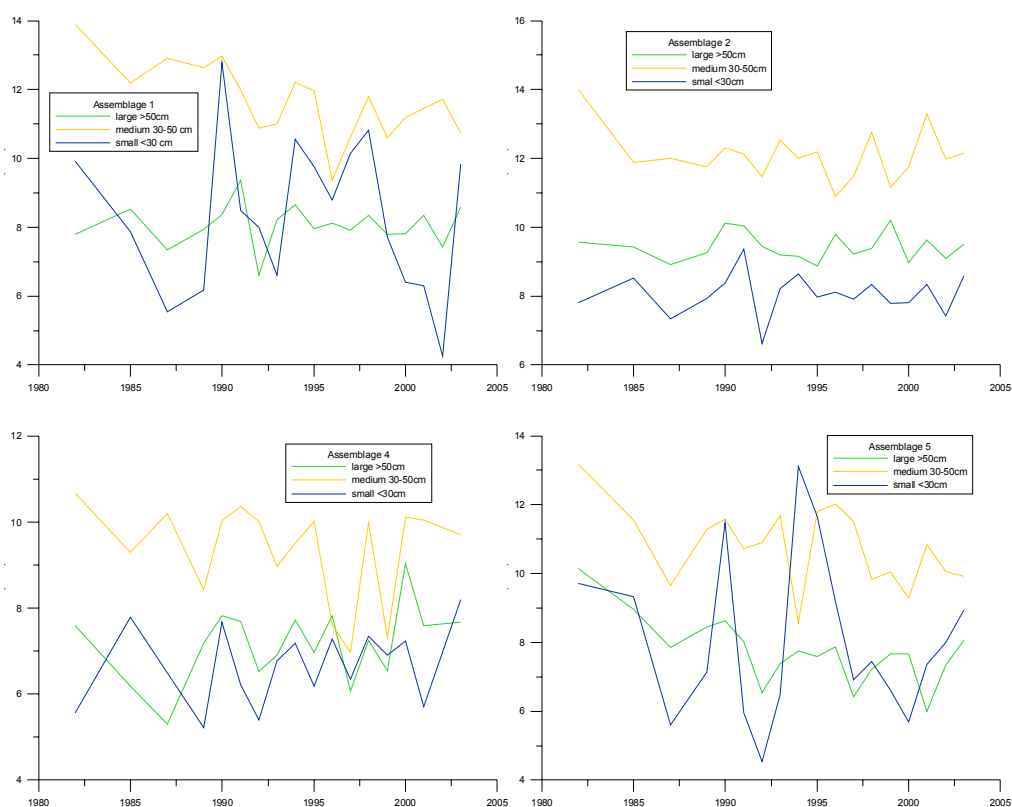


Figure 2.3.3.3. Change in abundance of small, medium and large fish over time.

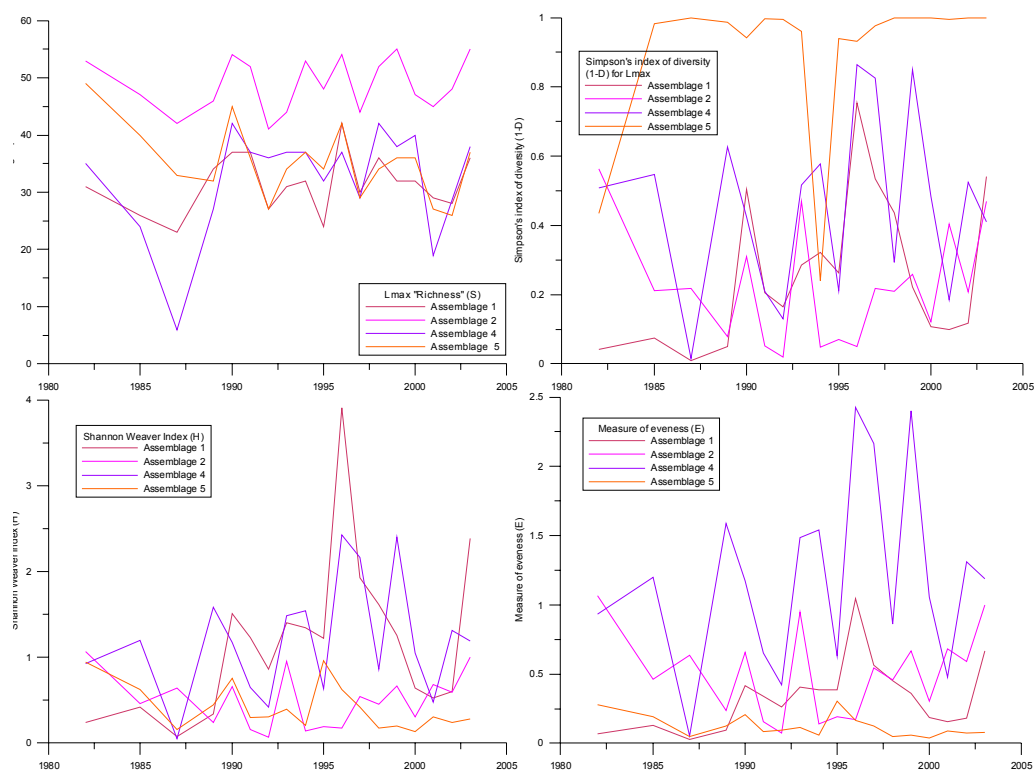


Figure 2.3.3.4. Trends in diversity indices of L_{\max} over time

Temporal patterns in trophic level

For these analyses we considered data from the Portuguese Continental region in the Atlantic worked out by IPIMAR's group. Survey data are available based on bottom trawl research sampling carried out annually on the shelf and slope up to a depth of 500–750m during summer and autumn in continental Portuguese waters since 1979 and are well described in Cardador *et al.* (1997). Only data from the autumn (fourth quarter) surveys for 1982, 1985 and 1987 and from 1989 to 2000 have been used in the present analysis.

During these surveys a total of 219 fish species were caught. Thirty nine species were excluded when one or more species characteristics were missing, these were all rare species, together representing less than 1% of the total number of all individuals in the data set. A further 2 species were excluded; these were snipefish *Macroramphosus sp.* and boarfish *Capros aper*. These small species (L_{\max} 30 and 20 cm respectively) have both become extremely abundant over the past decade (representing 76% and 8% of the total number of individuals in the study period). They are highly migratory species, with an area of occupancy that seems to be related with the shelf-edge path of the poleward current of warm water following bio-physical processes that are not yet fully understood (Peliz *et al.*, 2003).

The remaining 178 species were attributed a trophic level index directly from Froese and Pauly, 2000, which is defined as “the position in the food chain, determined by the number of energy- transfer steps to that level”.

For these analyses all the stations were separated into three geographic zones (north, centre and south) at 39.5 and 37.1 degrees latitude, and into depth strata (less than and more than 150m). This gave six assemblages to be used in this analysis. The criteria for these choices are largely taken from Gomes *et al.* (2001). For each assemblage and species the mean number of individuals per hour weighted by the number of sampled stations was calculated. Figure 2.3.3.5 shows the changes in trophic level and L_{\max} during 16 years (1984–2000) in the region of study.

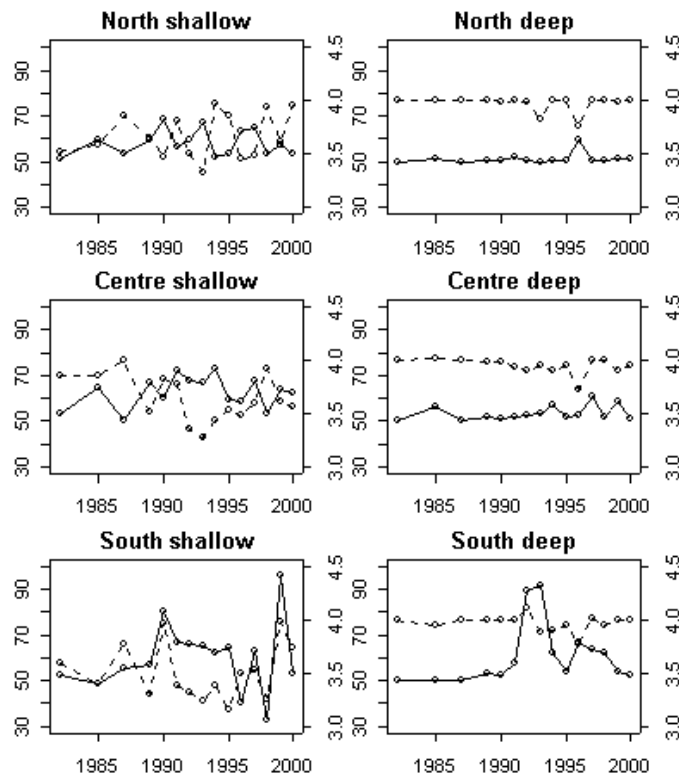


Figure 2.3.3.5. Changes in trophic level (dashed line) and L_{\max} (solid line) in six assemblages of the Portuguese Atlantic region

2.3.3.3 Discussion

Average L_{\max} , abundance metrics and diversity metrics did not show any consistent trends in any of the length-based fish assemblages. Overall, the metrics were relatively constant although some showed a slight increase in recent years.

When assemblages were classified according to depth and latitude it was found that all the shallow assemblages showed higher variabilities than the deep assemblages, both in trophic level and L_{\max} . This variability may be due to recruitment effects. Fish nurseries are situated on the shelf in shallower areas and fish species recruit in the Autumn (Borges, 1983; Borges, 1984, Cardador, 1995), according to the dynamics and timing of upwelling conditions in the Portuguese Western region (Santos *et al.*, 2001). L_{\max} was extremely variable, and seemed to respond to pelagic species (e.g., sardine and horse mackerel), and their year-class strength.

In the North and Central areas, trophic level and L_{\max} was rather constant. This might be explained by the decreasing effort in hours trawling per year of the Portuguese fish fleet during the same period (Hill *et al.*, 2001). In the Southern assemblage, these metrics have become more variable since 1991. A reason for that may be due to changes in faunal composition over time.

There are strong seasonal variations in environmental conditions in the study area, including upwellings that increase productivity. Species diversity in the area is relatively high, since Portugal is at the interface of the distribution of northern and southern species, and recent information has reported the presence of rare species from North Africa in the Algarve (Brander *et al.* 2003). Further studies examining the role of environmental conditions in structuring fish communities are required, and the potential for undertaking more regional studies along the continental shelf of the eastern North Atlantic should be investigated to better understand any potential distribution shifts in relation to oceanographic conditions.

2.3.4 Case study: Barents Sea (ICES Divisions I–II)

2.3.4.1 Introduction

Preliminary analyses of length-based community indicators for the Norwegian annual bottom trawl survey in the Barents Sea during January–March 1980–2003 have been made. Within this time series there were a number of known inconsistencies in the data, both regarding area coverage, gear specifications, and sampling protocol. Before analyses could be made, it was therefore necessary to modify the data in order to compensate for some of these changes. Here we present the modifications that were done and the resulting time series of four length-based measures of community composition

2.3.4.2 Material and methods

The area covered each year in these winter-surveys has varied due to varying distribution of drift-ice and due to lack of permission to enter foreign territorial waters in some years. Based on the actual coverage, a core-area was defined between 70–74°N and 15–35°E, within which the spatial distribution of hauls was reasonably similar in all years. The analyses were limited to this core area.

Gear changes included a change to Rockhopper ground gear in 1990, a reduction in the distance towed from three nm (up to and including 1985) to 1.5 nm thereafter, and several changes in vessels used. No compensation for these changes were made.

The survey was originally designed to focus on the commercially important species. Since then, there has been a gradual increased awareness of the importance of consistent recording of data for all species in the catches. It is believed though, that the total catches of each fish species or species group were recorded correctly throughout the time-series. However, in the first years length-distributions were not always recorded for species that were considered of minor importance. In many of these cases only the total weight of the species or species-group was recorded. To correct for this the catch number was estimated as catch weight divided by the mean individual weight of the species in all catches, all years combined, where both numbers and weight were available. In the same way, all missing length compositions were constructed from the total of all recorded length compositions of the species or species group, from all years combined. Since the mean over all years were used, these corrections will tend to reduce any trends that may exist throughout the time series.

Some species (e.g., eel-pouts) were also to a larger degree recorded as species groups in the early years of the time series. This makes length-based indices more suitable than, for example, diversity indices to examine changes over time.

Catch rates were expressed as numbers per 1 nautical mile trawling. All species were allocated to one of two groups (demersal and pelagic). The following indices were calculated for each year and for demersal and pelagic species separately:

- 1) Mean slope of the length-based size spectrum.
- 2) Mean weight across all species and hauls, weighted by the catch rates.
- 3) The mean length of all individuals in all catches
- 4) The 95th percentile length category was calculated from the total length composition of each species or species group. The mean across species and hauls was weighted by catch-rates.

2.3.4.3 Results

The slope parameter of the size spectrum for demersal species was very stable between years, and with no significant linear time trend (Figure 2.3.4.1). For pelagic species the slope was much more variable, but also this without any clear time-trend. Although the fitted linear regression was marginally significant ($p < 0.05$), this was mainly due to very few points in each end of the time series. Also the intercept showed variability without any global trends. The r^2 from the regressions of the size spectrums was reasonably high for all but a few regressions of pelagic species.

The three other measures, mean weight, mean length and mean 95th percentile length category, showed very similar patterns of variation in the demersal group, and no apparent correlation between the time series for pelagic and demersal species (Figure 2.3.4.2). There was no overall trend in these metrics but rather a periodicity of approximately seven years. For comparison, Figure 2.3.4.3 shows the variation in temperature in the central Barents Sea over the same

period. The periodicity is apparently similar and to some degree in phase with the community metrics of the demersal species. Further studies to examine the influence of environmental parameters on size-based indices of community structure are required.

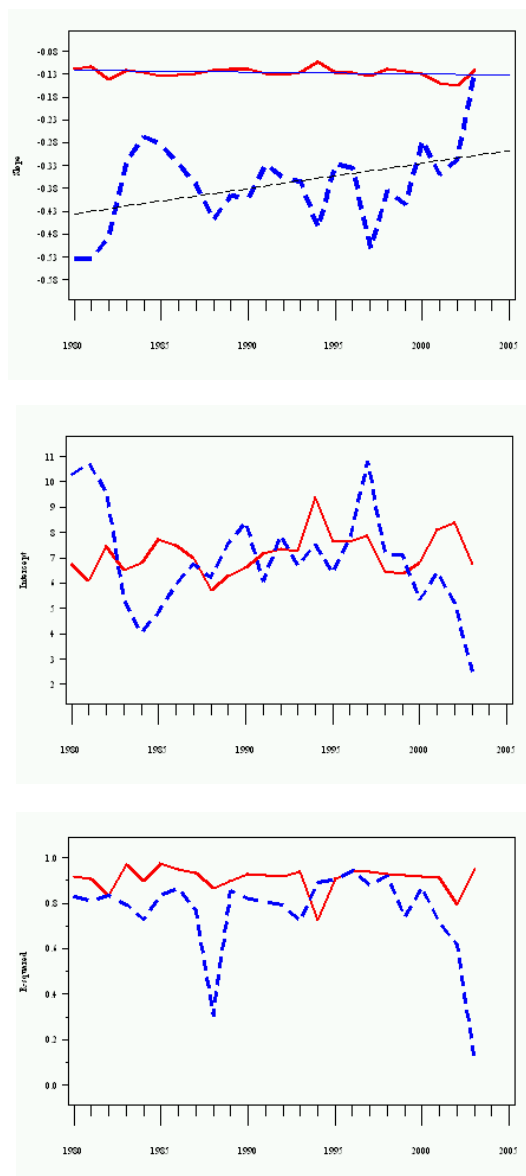


Figure 2.3.4.1. Slope parameter (upper), intercept (middle) and R^2 (lower) from linear regressions of the size spectra of catches of demersal (solid red line) and pelagic (broken blue line) species.

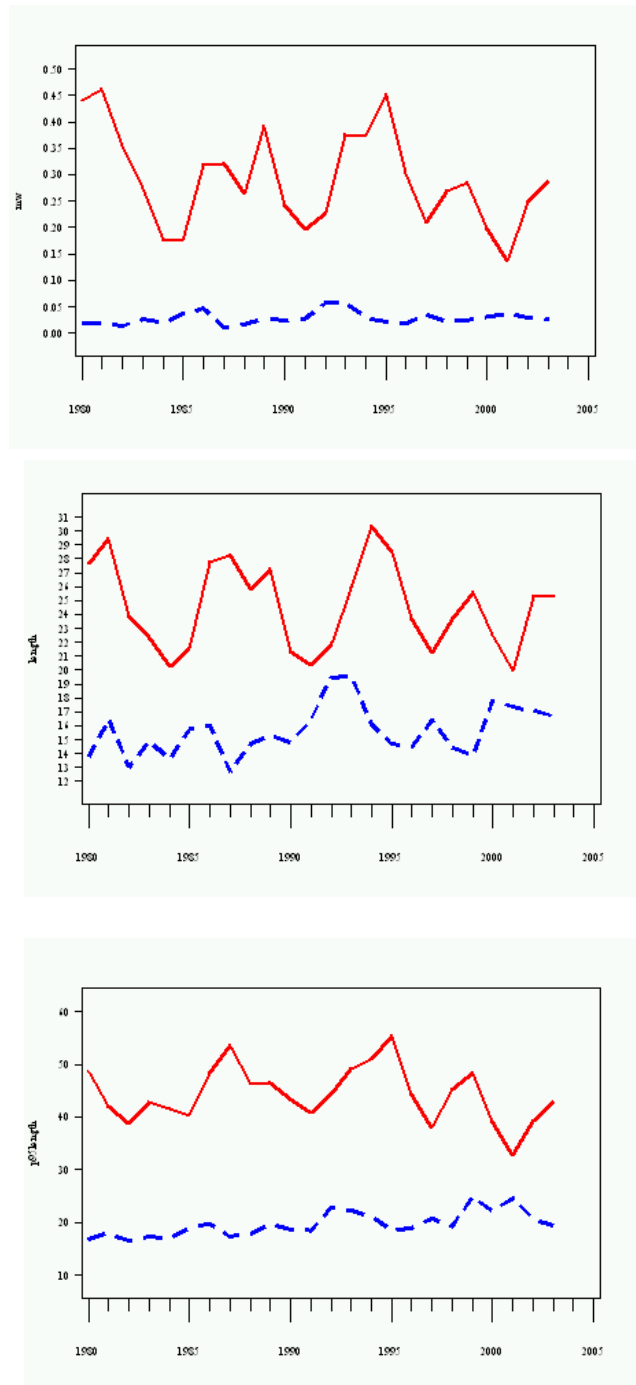


Figure 2.3.4.2. Mean individual weight (upper), mean length (middle) and mean 95-length-percentile of demersal (solid red line) and pelagic (broken blue line) species.

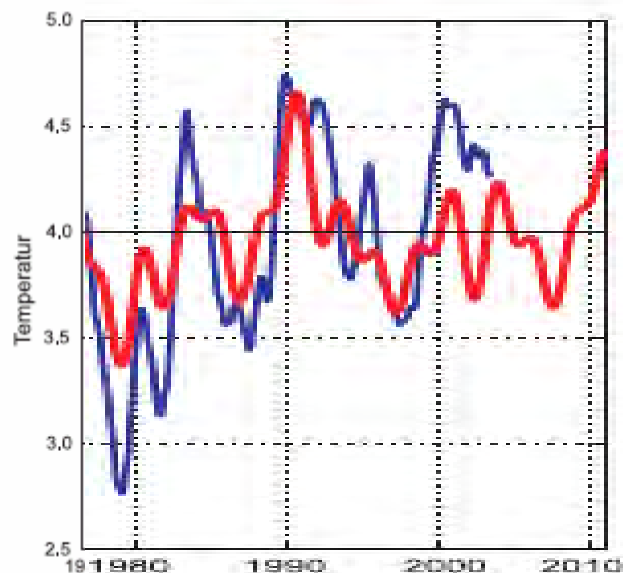


Figure 2.3.4.3. Observed and modelled temperature in the Kola section.

2.4 Exploring utility and application of EcoQOs over a range of spatial scales

2.4.1 Introduction

Different aspects of the utility and application of EcoQOs were evaluated for separate case studies at different spatial scales and these studies are reported below. Several issues were discussed: the applicability of different metrics derived from size spectra and their meaning and application to systems across the North Atlantic; spatio-temporal analyses of the effects of fishing on a range of fish community indicators using long-term temporal and spatial survey data and patterns in fishing intensity; and statistical power analyses of fish community indicators.

2.4.2 Evaluating EcoQO metrics and reference levels for the north-western North Sea

2.4.2.1 Introduction

Fish Communities is number 5 on the list of Ecological Quality (EcoQ) Issues requiring the setting of Ecological Quality Objectives (EcoQOs). Following the Statement of Conclusions issued by the Bergen Intermediate Ministerial Meeting in 1997, it has been anticipated that EcoQOs for this EcoQ issue would address multi-species, assemblage or community wide attributes. This has initiated a “hunt” for appropriate indicators of community “status”. This search has been particularly intensive with respect to fish communities since appropriate, spatially comprehensive groundfish survey data are available for many regions. In the North Sea, for example several internationally co-ordinated surveys covering most, if not all, of the North Sea area are carried out each year, and in many instances, these surveys have been carried out for several decades. In comparison, spatially and temporally comprehensive data are scarcer for benthic invertebrate communities.

There are many indices that convey information regarding a variety of community attributes (e.g., Washington 1984; Magurran 1988) that could easily be, or have been, applied to ground fish survey data (e.g., Greenstreet and Hall 1996; Greenstreet *et al.* 1999a; Jennings *et al.* 1999b; Jennings and Reynolds 2000; Jennings *et al.* 2002; Rogers *et al.* 1998; Rogers *et al.* 1999a, 1999b; Rogers and Ellis 2000). In an attempt to try and impose some order on this process, ICES WGECO proposed a set of *criteria* by which the effectiveness of different indices might be judged (ICES 2001, Table 2.4.2.1). After evaluation according to these criteria, WGECO concluded that metrics based on the mean size of fish, the proportion of large fish, and the ultimate body size of fish in the community, were the most appropriate indicators of the impact of fishing on fish communities (ICES 2001). These characteristics were subsequently adopted as the elements of EcoQ for Fish Communities in the Bergen Declaration in 2002.

Table 2.4.2.1. ICES criteria for a good EcoQO metric.

Criterion	Property
a	Relatively easy to understand by non-scientists and those who will decide on their use
b	Sensitive to a manageable human activity
c	Relatively tightly linked in time to that activity
d	Easily and accurately measured, with a low error rate
e	Responsive primarily to a human activity, with low responsiveness to other causes of change
f	Measurable over a large proportion of the area to which the EcoQ metric is to apply
g	Based on an existing body or time-series of data to allow a realistic setting of objectives

A problem with this approach is that these criteria concentrate almost entirely on demonstrating an unequivocal effect of an activity (e.g., fishing) on a particular indicator (e.g., mean size of fish in the community). Little emphasis is placed on the importance of the observed change in the community (e.g., a decrease in the size of fish in the community); whether it is of real ecological significance, requiring remedial action, or whether it is relatively trivial and might be ignored by managers. Application of these criteria focuses attention on the aspect of the community measured by the indicator that scores highest on most counts. A more positive approach might be to identify those aspects of community structure deemed to be most valuable, or most critical to ecosystem function, and then to select the most appropriate metric to monitor change in these attributes. The Convention on Biological Diversity (CBD), Agenda 21, and Annex V of OSPAR all place a legal obligation on the signatories to these agreements, which include the EC and all European states that border and exploit the biological resources of the North Sea, to conserve biodiversity and to restore biodiversity in degraded systems. No link has been clearly demonstrated between the mean size of fish in fish communities and community biodiversity. Knowing that the mean size of fish in the North Sea has declined does not necessarily help managers implementing an “ecosystem approach to management” in response to one of the most important “policy drivers” imposed on them to date. If the conservation and restoration of biodiversity is deemed of high importance, then the fish community indicators required by managers should incorporate indicators of biodiversity.

Biodiversity is, however, not the only issue on the agenda of an ecosystem approach to management. The various community indices in common use today quantify different aspects of the structure of communities and provide different levels of information regarding the functioning of communities within the broader ecosystem. Since the late 1990's there has been considerable scientific debate regarding the relationship between biodiversity and ecosystem function (Mooney, 2002; Huston and McBride, 2002; Emmerson and Huxham, 2002). Less diverse communities may well be less productive and less stable (Tilman and Downing, 1994; Tilman *et al.* 2002a, 2002b; Loreau *et al.* 2002; Petchy *et al.* 2002; Raffaelli *et al.* 2002). If so, then a reduction in fish assemblage species diversity brought on by over-fishing (e.g., Greenstreet and Hall 1996; Greenstreet *et al.* 1999) may hold major implications with respect to both the management of fisheries and to broader ecosystem management. Community-averaged life-history trait metrics convey information regarding the types of species present in a community, and these may also suggest changes in community/ecosystem function. In communities where average age at maturity has declined and the average growth rates have increased (e.g., Jennings *et al.* 1999b), increased productivity could be inferred. Where assemblage total biomass has remained relatively constant (Yang 1982; Sparholt 1990; Daan *et al.* 1990), but the biomass harvested has increased, such changes might imply increased resource cycling rates within the community. A holistic ecosystem approach to management needs to consider these other aspects of community structure and ecosystem function.

Rather than pursuing a quest for the “holy grail” indicator of indicators, identifying the principal issues of concern with respect to marine communities and ecosystems should be the immediate goal of marine scientists. The importance of biodiversity issues, and the policy drivers that require managers to address these, has already been described. Thus a clear need to apply biodiversity indicators to fish abundance data has been established. Fish size has been adopted as the element of EcoQ for fish communities in the Bergen Declaration, as it may be responsive to exploitation, and metrics that convey information on fish size are also necessary. Changes in productivity and resource cycling rates in fish communities is of potential concern. Indices that convey information of relevance to this issue may therefore also be desirable. Fishing down the trophic levels of marine food webs is another issue of potential concern (Beddington 1995; Pauly and Christensen 1995; Pauly *et al.* 1998). Metrics that indicate change in the trophic structure of fish communities would also provide information of value in the implementation of an ecosystem approach to management. Covering all these aspects of change in fish communities requires application of a suite of indicators.

In this example of the utility and application of community indicators to identify potential EcoQOs for fish communities, a suite of twelve indicators was applied to long-term ground fish survey data gathered from an area

approaching half the surface area of the North Sea situated in the north-western North Sea. To cover issues pertaining to the size of fish in the community three possible indicators were applied; the percentage of fish in the community exceeding a body-length of 30cm, the average weight of fish in the community, and the average $\text{Length}_{\text{infinity}}$ of fish in the community. To address issues related to community productivity, changes in the life-history characteristics of fish in the community were examined. Three possible indicators were applied: the average age and the average length at maturity, and the average growth rate of fish in the community. To examine trends in fish community biodiversity, five potentially useful indicators were applied; the number of species in samples, Margaleff's index of species richness, Pielou's index of species evenness, and Hill's (1973) N1 and N2 indices of species diversity. Finally, to explore possible changes in the trophic structure of the fish community, trends in the average Nitrogen stable isotope ratio were examined.

Previously, when the ICES criteria for a good indicator were applied to these metrics, several scored relatively poorly (ICES 2001). Demonstrating a clear link between changes in human activity and performance of the indicators was one of the major difficulties encountered. Thus several of these indicators fared badly against criteria b, c, and e (see Table 2.4.2.1). WGFE is primarily concerned that, in implementing an ecosystem approach to management in the North Sea, the detrimental impacts of fishing are minimised. In applying these indices to fish abundance data therefore, an analytical design is adopted such that specific hypotheses related to variation in fishing activity are tested. In this way the causal link between the human activity and index performance should be more fully established, and reservations regarding these three criteria reduced.

Recent work has led to the proposal of four *a priori* hypotheses as to how some characteristics of the groundfish species assemblage in the North Sea may be affected by fishing (Bianchi *et al.* 2000; Denney *et al.* 2002; Duplisea *et al.* 1997; Gislason and Rice 1998; Greenstreet and Hall 1996; Greenstreet *et al.* 1999a; Jennings *et al.*, 1998; Jennings *et al.* 1999b; Jennings *et al.* 2001; Jennings *et al.* 2002; Jennings and Mackinson 2003; Pinnegar *et al.* 2002):

- The proportion of large fish, mean weight of fish, and average ultimate body-length of fish in the groundfish assemblage should be lower in areas most disturbed by fishing;
- The species richness and species diversity of the groundfish assemblage should be lower in areas most disturbed by fishing;
- The life-history characteristics of the groundfish assemblage should change, growth rates should be highest, and size and age at maturity should be lowest, in areas most disturbed by fishing;
- The trophic level at which fish belonging to the groundfish assemblage feed should be lower in areas most disturbed by fishing.

These hypotheses underpin the analytical design. All three predictions follow from first order effects of fishing as a source of mortality that is not equal across all species and sizes of fish in the community. More complex ecological processes, such as inter-specific competition, top-down/bottom-up control, resource supply and cycling, trophic level transfer efficiency, and productivity (Connell 1975; 1978; Paine 1974; Huston 1994; Pauly and Christensen 1995) could all serve to amplify these changes. Scottish August Groundfish Survey (SAGFS) data, and international fishing effort data, and information on life-history characteristics and trophic level of the species encountered in the SAGFS are analysed to test these hypotheses. Previous studies have adopted a pseudo-correlative approach, comparing time-series trends. For example, comparing temporal trends in species diversity (Greenstreet *et al.* 1999a), or assemblage-averaged growth rate (Jennings *et al.* 1999b), in a region where fishing effort has increased over time. However, a major drawback of the correlative approach is its inability to definitely confirm cause and effect. Thus, whilst these studies might suggest that fishing has caused changes in the fish community, they cannot definitely confirm the link. In this study, spatial and temporal analyses are combined in an attempt to strengthen the case that the changes observed were in fact caused by fishing.

Actually setting EcoQOs for each of these metrics, in other words, what is the metric value that managers should aim for, remains a significant obstacle to utilising and applying these metrics within an EcoQO framework. EcoQ was defined as the "Ecological quality of ecosystems is an overall expression of the structure and function of the aquatic systems, taking into account the biological community and natural physiographic, geographic and climatic factors as well as physical and chemical conditions including those resulting from human activities". EcoQOs were defined as "the desired level of EcoQ relative to a reference level" where reference levels were defined as "the level of the EcoQ where the anthropogenic influence on the ecological system is minimal". The first major problem lies in establishing reference levels, the metric value expected in the absence of the human activity. The second difficulty is deciding the metric level to aspire to that is consistent with good ecosystem governance, yet still permits the continuance of a viable fishing industry. The identification of appropriate reference levels is primarily a question that should be addressed to marine scientists; the second is very much a political question with significant social implications. Accordingly, the

analytical design adopted here is intended to address the problem of identifying reference levels for each of the community metrics applied to the fish community of the north-western North Sea

2.4.2.2 Methods

Analytical Design

Data for 75 ICES rectangles, divided into three groups, or treatments, of low, medium and high “current” fishing effort were analysed. Twelve potential indicators of different characteristics of the ground-fish assemblage were examined to determine the extent to which each was affected by fishing. The hypothesis that each metric was “most affected” in the rectangles of highest fishing disturbance, and least affected in the rectangles of lowest disturbance was tested. Even should the data support these initial hypotheses, this does not necessarily confirm that fishing has been responsible for any observed differences. The alternative interpretation, however unlikely, that fishing activity may have been attracted to areas where the ground-fish assemblage may have displayed particular characteristics, such as low average weight, or low species diversity, cannot at this point be discounted. To rule out this alternative interpretation, long-term time-series trends for each of the groups of rectangles were examined. If fishing is responsible for the change in the community characteristics, then predictable temporal trends should be apparent. Little or no long-term trend should be apparent in rectangles where fishing disturbance is low, whereas in rectangles affected by fishing, temporal trends in a predictable direction should be detected. The greater the impact from fishing, the steeper the gradient should be (Figure 2.4.2.1). An assumption underlying this analytical design is that prior to any apparent fishing effects, the community characteristics in the different treatment areas had the same start point, and that each was subjected to approximately the same low level of fishing disturbance. This is examined in a later section dealing with reference points and the setting of target values.

In adopting this analytical design, we have attempted to follow, as far as was possible, a one-way ANOVA design. However, it is important to realise that the distribution of fishing effort was not random across the 75 rectangles (Jennings *et al.* 1999a; Greenstreet *et al.* 1999b, Figure 2.4.2.2). A true ANOVA design would have had each “treatment” set of rectangles distributed randomly across the 75 rectangles. This has two major implications.

- Spatial variation could introduce a potentially confounding effect. Concentration of the impact of fishing into restricted areas could magnify the effect of fishing on the demersal fish community. Nevertheless this is still a fishing effect. It will lead to similar distribution in the community characteristic being investigated. The question is, can this spatial factor introduce the sort of trends we anticipate independently of fishing?
- Spatial cohesion of both the “treatment” and the “effect” could, through spatial auto-correlation, reduce the independence of the data. This has consequences with respect to estimation of the actual degrees of freedom in any statistical analysis. While we have presented significance levels for the ANOVA results, some caution is necessary in interpreting these. Analysis of ground-fish survey data collected at high spatial resolution (25 to 30 half-hour GOV samples collected within a 20km by 20km area) suggests that auto-correlation between species abundance is almost entirely diminished at distances of around 10 to 15km. Examination of variograms for each of the community characteristics suggested that spatial auto-correlation was diminished over a distance of around 100 to 150km (two to three ICES rectangles), suggesting that the true number of total degrees of freedom may be as few as 25 to 40.

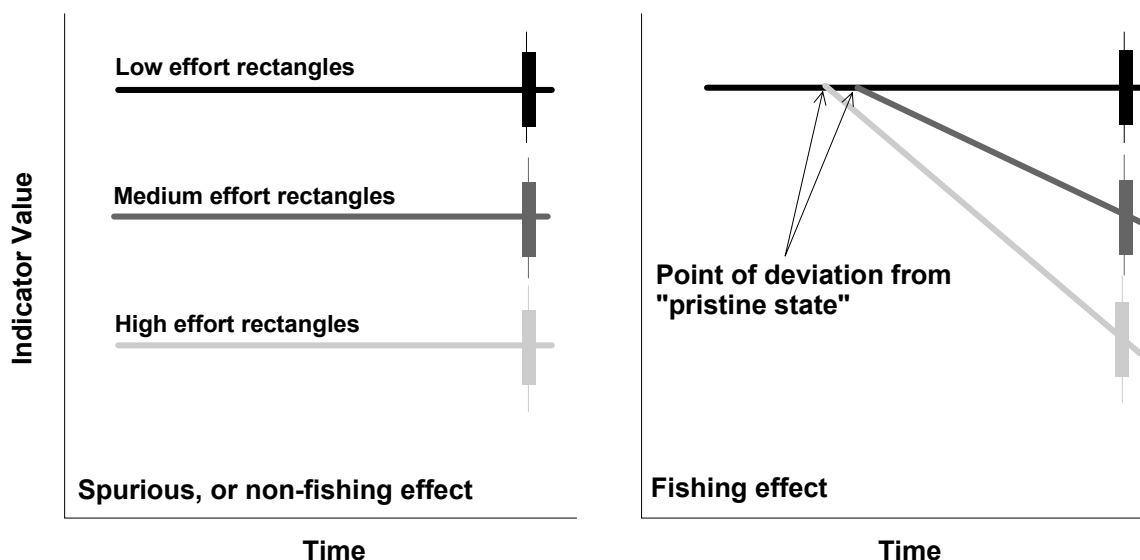


Figure 2.4.2.1. Illustration of the analytical design to test specific hypotheses. Box and Whisker plots to the right of each panel indicate community indicator values in each fishing effort treatment in the contemporary period. The lines show the long-term temporal trajectories of indicator values to reach this point under circumstances where fishing activity explains the difference between the treatments and under circumstance where the difference is not a fishing effect.

Ground-fish survey data

Scottish August Ground-fish Survey (SAGFS) data collected in 75 ICES statistical rectangles located in the north-western North Sea where survey coverage was most complete were examined (Figure 2.4.2.2). Only trawl samples collected using a 48-foot Aberdeen Otter Trawl towed for one hour were included in the data set. Only data for those ground-fish species likely to be well sampled by the gear were analysed. Pelagic species, such as herring, sprats and sandeels, were all excluded. The results therefore only apply to the demersal ground-fish community occupying the area. For more details regarding the data see Greenstreet and Hall (1996) and Greenstreet *et al.* (1999a).

To determine “contemporary” levels of each of the community indicators for each rectangle, species at length abundance data covering a period of 14 years from 1983 to 1996 were extracted. All samples in this data extraction were collected by the same survey vessel, FRV *Scotia (II)*. For one rectangle only ten trawl samples were available. This rectangle was not sampled in 1983, 1985, 1987 or 1995. To avoid sample size dependency problems, sampling effort was standardised to ten trawls in the other 74 rectangles by excluding, as necessary, trawl samples selected at random from these four years. Previous analysis of SAGFS data has indicated that it is necessary to aggregate at least five one-hour trawl samples in order to derive reliable community metrics. All ten trawl samples in each rectangle were therefore combined to provide a single aggregated, highly standardised, species abundance sample for each rectangle upon which to calculate each community indicator.

For the second analysis, looking at “long-term temporal trends” in each of the groundfish assemblage indicators in rectangles varying in the level of fishing effort to which they had been subjected, data from the full time-series, spanning the period 1925 to 1996, were used. Following Greenstreet *et al.* (1999a), data were pooled into “groups” of two or three years to ensure adequate sampling effort in each time-period/effort “treatment” cell.

Fishing effort data

International fishing effort (hours fished) for the period 1990 to 1995 were used to define rectangles subject to three different levels of fishing disturbance (Jennings *et al.* 1999a; Jennings *et al.* 2000). Total annual average fishing effort across the 75 statistical rectangles amounted to 963,216 hours of fishing, 67% of which consisted of otter trawling, 12% beam trawling and 21% seine netting. Otter trawl is the predominant gear used in this part of the North Sea (Greenstreet *et al.* 1999b). The treatment effort levels set were therefore based solely on this gear. Average annual effort values were calculated to provide estimates of the “current” spatial distribution of fishing effort across the 75 ICES statistical rectangles for which groundfish survey data were available. These ranged from 645 h.yr⁻¹ to 63,794 h.yr⁻¹. Three broad categories (“treatments”) were defined; 40 rectangles of relatively low otter trawling intensity, from 0 to 4999 h.yr⁻¹, 25 rectangles of medium otter trawl effort, from 5000 to 19,999 h.yr⁻¹, and 10 rectangles of high otter trawling intensity, exceeding 20,000 h.yr⁻¹. The distribution of rectangles belonging to each of these treatments is shown in Figure 2.4.2.2.

Community indicators

Twelve potential indicators of “the health” of the demersal fish community were examined. These were broadly divided into four groups, or types, of indicator: indicators of the mean size of fish in the community; indicators of the average life history characteristics of the fish making up the community; indicators of community species diversity; and an indicator of trophic structure in the fish community.

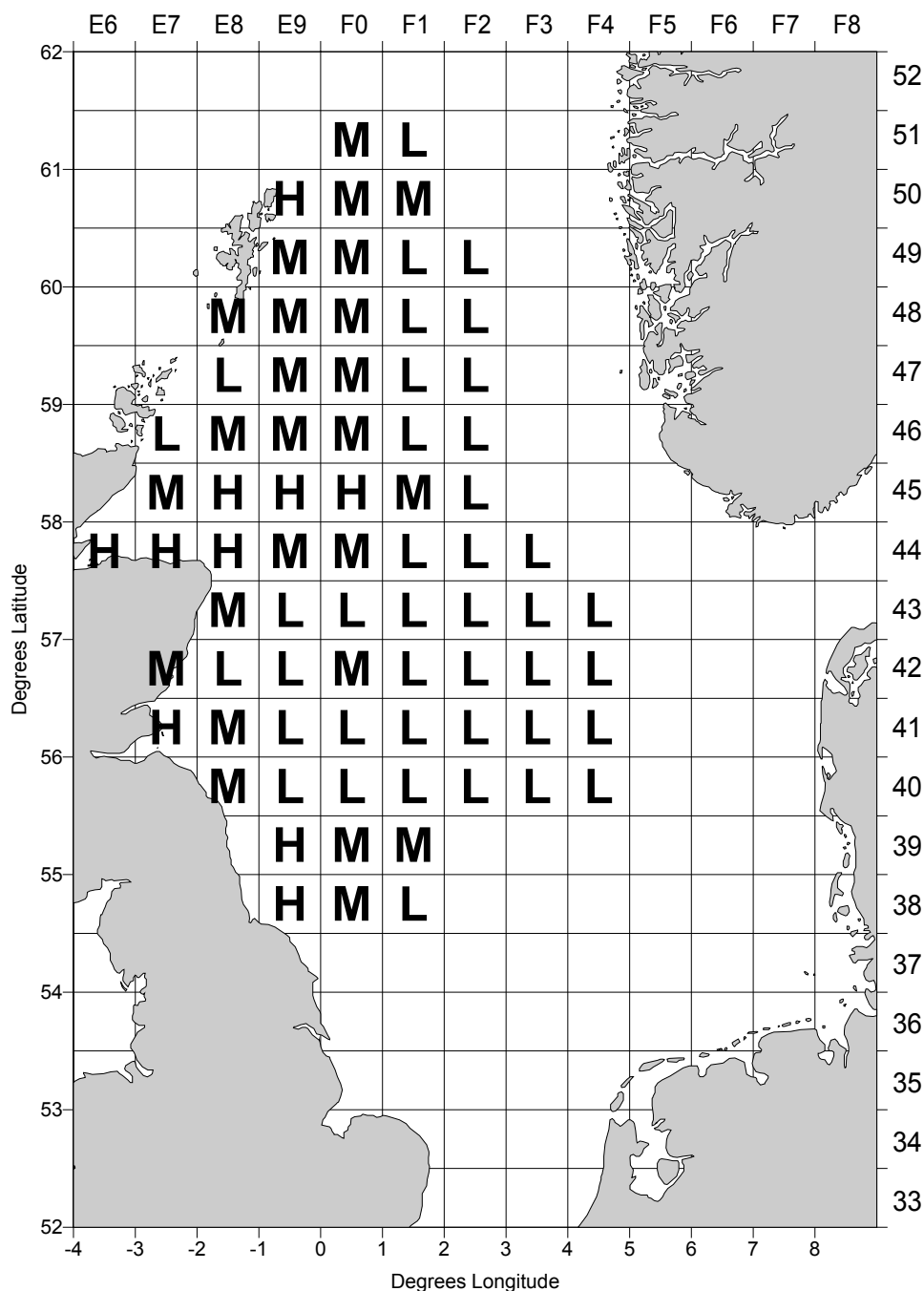


Figure 2.4.2.2. Chart of the North Sea showing the 75 ICES statistical rectangles for which SAGFS data were available for analysis and indicating the effort level treatment to which each rectangle was assigned.

Size-composition indicators

Species abundance at length data in the recent effort standardised database were examined to determine an appropriate length at which a fish might be considered to be large. 95% of all the individual fish recorded were 30cm in length or less. The top five percentile of the length distribution, fish over 30cm in length, was therefore defined as “large”. Length-weight relationships were used to convert abundance at length to weight at length. Average weight in a

treatment was determined simply by dividing the total weight in the sample by the total number of individuals. Length_{Infinity} data, determined from the von Bertalanffy growth equation calculated for each species, were available for 28 of the 56 species included in the SAGFS database (Table 2.4.2.2) (Jennings *et al.*, 1998; Jennings *et al.*, 1999b). These 28 species accounted for over 98% of the individuals sampled by the SAGFS in any of the spatial/temporal “treatments”. Species abundance data were converted to the number of individuals with particular Length_{Infinity} values, and the mean value for each spatial/temporal “treatment” was computed. Length_{Infinity} could also be considered to be a life-history characteristic and used as an indicator in this context as well (e.g., Jennings *et al.*, 1999b).

Life-history characteristic indicators

Information regarding three life-history characteristics, growth rates, Age_{Maturity}, and Length_{Maturity}, were available for 28 of the 56 species included in the SAGFS database (Table 2.4.2.2). The growth rate data used were the parameter values determined from the von Bertalanffy growth equation calculated for each species. The von Bertalanffy growth parameter is not strictly a rate value, but is used here as an index equivalent to growth rate. Age_{Maturity} and Length_{Maturity} values were determined by observation, either from recent survey data, or with recourse to the literature (Jennings *et al.*, 1998; Jennings *et al.*, 1999b). These 28 species accounted for over 98% of the individuals of demersal fish species sampled by the SAGFS in any of the spatial/temporal “treatments”. Species abundance data were converted to the number of individuals with particular characteristic values, and the mean value for each characteristic for each spatial/temporal “treatment” computed.

Table 2.4.2.2: List of species for which life-history character information was available.

Common name	Scientific name	Len _{In}	GR	Age _{mat}	Len _{mat}
Spurdog	<i>Squalus acanthias</i>	90.20	0.15	6.50	66.80
Lesser spotted dogfish	<i>Scyliorhinus canicula</i>	90.00	0.20	5.00	58.00
Starry ray	<i>Amblyraja radiata</i>	66.00	0.23	4.00	45.70
Cuckoo ray	<i>Leucoraja naevus</i>	91.64	0.11	9.00	58.82
Spotted ray	<i>Raja montagui</i>	97.80	0.15	6.00	66.60
Cod	<i>Gadus morhua</i>	123.10	0.23	3.80	69.70
Haddock	<i>Melanogrammus aeglefinus</i>	68.30	0.19	2.50	33.50
Whiting	<i>Merlangius merlangus</i>	42.40	0.32	1.50	20.20
Saithe	<i>Pollachius virens</i>	177.10	0.07	4.60	55.40
Norway pout	<i>Trisopterus esmarki</i>	22.60	0.52	2.30	18.60
Poor cod	<i>Trisopterus minutus</i>	20.30	0.51	2.00	13.02
Torsk	<i>Brosme brosme</i>	88.60	0.08	9.60	49.55
Four-bearded rockling	<i>Enchelyopus cimbrius</i>	36.00	0.20	3.00	14.00
Hake	<i>Merluccius merluccius</i>	103.60	0.11	3.00	36.90
Anglerfish	<i>Lophius piscatorius</i>	135.00	0.18	5.00	75.00
Grey gurnard	<i>Eutrigla gurnardus</i>	46.16	0.16	2.50	20.95
Bull rout	<i>Myoxocephalus scorpius</i>	34.00	0.24	2.00	15.00
Hooknose	<i>Agonus cataphractus</i>	17.40	0.42	2.00	9.22
Catfish	<i>Anarhichas lupus</i>	117.40	0.05	6.00	42.50
Dragonet	<i>Callionymus lyra</i>	22.20	0.47	1.50	13.29
Megrim	<i>Lepidorhombus whiffiagonis</i>	51.80	0.07	2.60	19.15
Turbot	<i>Psetta maxima</i>	57.00	0.32	4.50	46.00
Witch	<i>Glyptocephalus cynoglossus</i>	45.50	0.16	3.00	20.00
Long rough dab	<i>Hippoglossoides platessoides</i>	24.60	0.34	2.60	15.14
Halibut	<i>Hippoglossus hippoglossus</i>	204.00	0.10	5.80	83.00
Common dab	<i>Limanda limanda</i>	26.70	0.26	2.25	13.08
Lemon sole	<i>Microstomus kitt</i>	37.10	0.42	4.00	27.00
Plaice	<i>Pleuronectes platessa</i>	54.40	0.11	2.50	26.60

Species richness and diversity indicators

Species diversity conceptually consists of two different aspects of species relative abundance: firstly the actual number of species included in any particular sample, and secondly the evenness of the distribution of individuals between the species encountered. Here we used five different indices each differing in the extent to which they are influenced by one or other of these two aspects of species diversity. Species richness was simply the count of all species encountered in the aggregated samples. This index is highly sample size dependent, thus we also applied an index that takes account of the number of individuals included in the sample, Margaleff's species richness index, d , determined as $(S-1)/\ln(N)$ where S is the number of species and N the number of individuals in the sample. As a straight indicator of evenness we

applied Pielou's index, J , given as $H/\ln(S)$, where H is the Shannon-Weiner index computed for the sample, determined as $-\sum_i p_i \ln p_i$ where p_i is the proportion of the total sample contributed by the i th species (Magurran 1988), and S is the number of species recorded in the sample. Two diversity indices, Hill's (1973) N1 and N2, were also computed for each of the 75 rectangles' aggregated samples. Hill's N1 diversity index is the exponential of the Shannon-Weiner index, and is thus, $e^{-\sum_i p_i \ln p_i}$. N2 is the reciprocal of Simpson's index, D , where D is computed as $\sum_i p_i^2$ where p_i is again the proportion of the total sample contributed by the i th species (Magurran 1988). Thus N2 is $1 / \sum_i p_i^2$. N1 is more sensitive to the number of species recorded in the sample, where as N2 is more sensitive to the evenness of the distribution of individuals between species.

Trophic level indicators

Information regarding the trophic level at which fish were feeding were available for 26 of the 56 species included in the SAGFS database (Jennings *et al.* 2001; Jennings *et al.* 2002). Variation in the trophic level at which fish were feeding was estimated by determining the stable Nitrogen isotope ratios present in the white muscle tissue of fish sampled throughout the North Sea and Celtic Sea. Increase in the $N_{15}:N_{14}$ ratio (henceforth referred to as the Nitrogen Ratio) reflects a higher trophic level diet (Minawaga and Wada 1984). Relationships for Nitrogen Ratio at length were determined for a total of 31 species, of which 26 were encountered in the SAGFS database (Table 2.4.2.3). These 26 species accounted for over 98% of the individuals sampled by the SAGFS in any of the spatial/temporal "treatments". Species abundance at length data were converted to the number of individuals with given Nitrogen Ratio values, and the mean value for each spatial/temporal "treatment" was computed.

2.4.2.3 Results

Box and Whisker plots of the contemporary values of each metric in rectangles of high, medium and low fishing effort are shown in Figure 2.4.2.3 and the results of one way ANOVA analysis are given in Table 2.4.2.4. With the exception of the $\text{Length}_{\text{maturity}}$ and Nitrogen Ratio indicators, all ANOVAs were significant at $P < 0.05$ or lower. For these two indicators, no obvious effect of fishing was apparent in the contemporary data. For all other indicators, significant differences between fishing treatments were observed. These differences were in the direction anticipated by the stated hypotheses, except for the two species richness indices. Both species richness metrics indicated no difference between the high and medium fishing effort treatments, whilst in rectangles with low fishing effort, species richness was significantly lower. Where the results were in the predicted direction, two different effects were observed. Thus gradients across the fishing effort treatments were apparent in the percentage of large fish, average fish weight, and average age at maturity in the community: low values were observed on high fishing effort rectangles, intermediate values on medium effort rectangles, and the highest values on low fishing effort rectangles. In all other instances where variation in the indicators supported the hypotheses (the three diversity indices and average $\text{Length}_{\text{infinity}}$), no difference was detected between high and medium effort treatments, but both differed from the low effort treatments.

Table 2.4.2.3. Parameters and test statistics for linear relationships between length (L mm log₁₀ transformed) or weight (W g log₂ transformed) and $\delta^{15}\text{N}_{0/00}$ or estimated trophic level (T.L.) of North Sea fishes. The form of the fitted relationships is $\delta^{15}\text{N}_{0/00} = a + b(\log_{10} L)$. From Jennings *et al.* (2002).

Species	Length (log ₁₀) vs $\delta^{15}\text{N}_{0/00}$ and T.L.						
	$\delta^{15}\text{N}_{0/00}$		T.L.				
	a	b	a	b	r ²	F	p
Wolfish	19.52	-2.53	6.08	-0.74	0.25	1.4 _{1,4}	0.308
Scaldfish	13.47	0.96	4.31	0.28	0.32	2.3 _{1,5}	0.190
Solenette	10.49	2.42	3.43	0.71	0.63	20.1 _{1,12}	<0.001
Dragonet	11.75	0.33	3.80	0.10	0.03	0.1 _{1,13}	0.848
Four bearded rockling	0.10	6.94	0.37	2.04	0.46	12.8 _{1,15}	0.003
Grey gurnard	10.40	1.93	3.40	0.57	0.08	1.5 _{1,17}	0.234
Cod	7.30	3.17	2.49	0.93	0.18	4.9 _{1,22}	0.037
Witch	-7.38	7.92	-1.83	2.33	0.37	5.3 _{1,9}	0.046
Long rough dab	10.82	0.95	3.53	0.28	0.02	0.4 _{1,19}	0.550
Megrim	-8.82	7.90	-2.25	2.32	0.81	37.7 _{1,9}	<0.001
Dab	9.74	2.04	3.21	0.60	0.06	2.1 _{1,30}	0.159
Anglerfish	0.67	4.88	0.54	1.44	0.60	14.9 _{1,10}	0.003
Haddock	8.63	2.24	2.88	0.66	0.20	7.7 _{1,30}	0.010
Whiting	4.07	4.99	1.54	1.47	0.41	5.5 _{1,8}	0.047
Hake	7.20	2.30	2.46	0.68	0.17	1.2 _{1,6}	0.318
Lemon sole	6.04	3.10	2.12	0.91	0.14	1.6 _{1,10}	0.236
Plaice	24.01	-3.89	7.43	-1.15	0.35	10.8 _{1,20}	0.004
Saithe	-2.32	5.53	-0.34	1.63	0.71	60.0 _{1,23}	<0.001
Cuckoo ray	-3.37	6.12	-0.65	1.80	0.78	57.6 _{1,16}	<0.001
Starry ray	0.88	5.16	0.60	1.52	0.69	28.4 _{1,13}	<0.001
Lesser spotted dogfish	6.97	2.34	2.39	0.69	0.03	0.14 _{1,4}	0.726
Norway haddock	-5.79	7.71	-1.36	2.27	0.87	32.1 _{1,5}	0.002
Sole	2.20	5.48	0.99	1.61	0.32	10.8 _{1,23}	0.003
Spurdog	-54.87	22.97	-15.80	6.76	0.90	34.0 _{1,4}	0.004
Norway pout	1.51	5.25	0.79	1.54	0.53	18.0 _{1,16}	0.001
Poor cod	21.28	-2.61	6.60	-0.77	0.06	1.0 _{1,15}	0.342

Table 2.4.2.4. One way ANOVA analysis results comparing effort treatments for each community metric in the contemporary period.

Indicator	ANOVA P value	Tukey post-hoc comparison P values			Ranking Order
		High v Medium	High v Low	Medium v Low	
Percentage of large fish	0.003	0.294	0.005	0.063	H<M<L
Average weight of fish	0.041	0.319	0.038	0.381	H<M<L
von Bertalanffy Length _{infinity}	0.007	0.969	0.130	0.009	H=M<L
Age at maturity	0.001	0.273	0.022	0.022	H<M<L
Length at maturity	0.108	0.997	0.317	0.142	H=M=L
von Bertalanffy growth parameter	0.000	0.903	0.022	0.000	H=M>L
Number of species	0.000	0.713	0.024	0.000	H=M>L
Margaleff's species richness index	0.000	0.689	0.085	0.000	H=M>L
Pielou's species richness index	0.000	0.974	0.019	0.000	H=M<L
Hills' N1 species diversity index	0.005	0.950	0.129	0.006	H=M<L
Hills' N2 species diversity index	0.045	0.972	0.303	0.050	H=M<L
Nitrogen stable isotope ratio	0.199	0.979	0.389	0.271	H=M=L

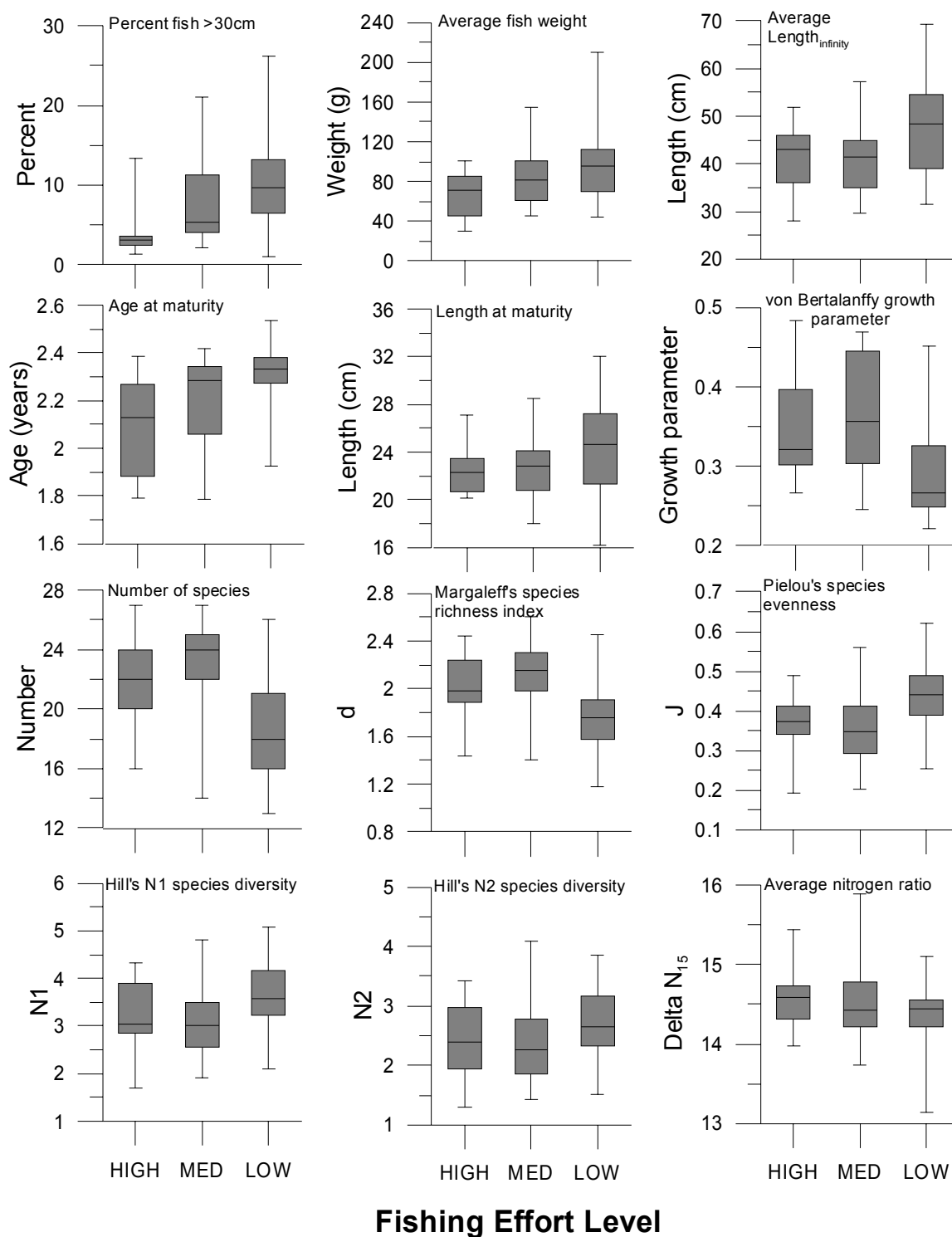


Figure 2.4.2.3. Box and Whisker plots showing fish community metric values in three fishing effort treatments in the contemporary period.

Long-term trends for the twelve community metrics were determined for each of the three fishing effort treatments. Trends for the three fish size metrics are shown in Figure 2.4.2.4, for the three life history characteristics in Figure 2.4.2.5, for the five species richness and diversity indices in Figure 2.4.2.6, and for the trophic structure metric in Figure 2.4.2.7. Regression analysis statistics for these plots are provided in Table 2.4.2.5. In two instances, significant long-term trends were observed in rectangles with low fishing effort. Thus, even in rectangles where fishing activity was low average growth rates in the fish community increased over the period 1925 to 1996. Similarly, species richness, as indicated by Margaleff's index, declined significantly in the low fishing effort rectangles. These results were not anticipated by our hypotheses, however as predicted, the gradients of the trend lines for both indicators were steeper in the medium and high fishing effort treatments. Long-term trends in the Nitrogen Ratio were not significant in any of the

three fishing effort treatments. For all other indicators, statistically significant trends in the directions predicted by the hypotheses were observed in the high fishing effort treatments, and in all cases, again as predicted, the gradients of the slopes were steeper than the trend-line gradients fitted to the low effort treatments. Trends in the percentage of fish >30cm and species richness (count of species) were both not statistically significant in the medium fishing effort treatment. However, for all remaining metrics, and in line with the hypotheses' predictions, significant trends were detected on the medium effort treatments, all with steeper trend-line gradients than those fitted to the low effort treatments.

With some exceptions therefore, the potential fish community indicators examined here have behaved in the way predicted by the initial hypotheses. Gradients of the long-term trend-lines in the low effort rectangles have either not differed significantly from a slope of zero, or where significant trends were detected, the gradients were shallower than those in the high and medium effort treatments. These two exceptions are discussed below, but for simplification, in the next step of the analysis the two metrics involved, Margaleff's species richness index and the von Bertalanffy growth parameter, are treated as if their long-term trends were zero. These results suggest that the low effort treatments provide an indication of the situation "where the anthropogenic influence on the ecological system is minimal". Under these circumstances, the mean and the 95% confidence limits of the low effort treatment data may be used as "reference levels" against which EcoQOs might be set.

Figure 2.4.2.8 shows these "reference levels" for all the twelve community metrics, and against these, the trend lines for the medium and high effort treatments have been plotted. These plots suggest that the SAGFS data set is sufficiently long-lived as to have "captured" the point in time where fishing activities in the north-western North Sea started to affect the characteristics of the fish community examined. Figure 2.4.2.9 shows the same "reference levels" as Figure 2.4.2.8, but here the actual time series data for the high fishing effort rectangles are shown. These plots suggest that with respect to the fish size metrics, since 1970 the fish community in the most heavily fished parts of the area was outside the "reference level" lower 95% confidence limit for over 95% of the time. An almost identical situation is apparent for the three life-history metrics.

At first glance the situation does not seem to be so bad with respect to fish community species richness and species diversity. Since 1970, only around half the data points for the five metrics fall below the "reference level" lower 95% confidence limit. However, the trend-lines shown in Figure 2.4.2.8, and the data in Figure 2.4.2.9, suggest a problem with the original assumptions illustrated in Figure 2.4.2.1. It is clear that species richness and diversity in the high and medium effort rectangles at the start of the SAGFS period were actually higher than those in the low fishing effort rectangles. Thus the assumption that the three effort level treatment trend-lines have deviated away from a similar start point by differences in fishing effort levels is violated. It would seem that levels of fishing activity have been greatest in areas of high fish species richness and diversity, and that fishing activity in these areas has reduced fish species richness and diversity to the same, or even lower, levels found in rectangles of low fishing effort. Under these circumstances, the mean metric values in the low effort treatment do not adequately represent the non-anthropogenically influenced situation. The reference level lines for these metrics in figures 2.4.2.8 and 2.4.2.9 should in fact be substantially higher, in which case the situation since 1970 with respect to species richness and diversity would almost certainly be similar to the fish size and life-history characteristic metrics.

With respect to changes in the trophic structure of the fish community in the north-western North Sea, variation in the Nitrogen Ration would appear to suggest that this has been unaffected by variation in fishing activity.

Table 2.4.2.5. Long-term temporal trends regression analysis results for each community metric in three fishing effort treatments

Indicator	Effort	Constant	Slope	P	Outlier excluded	N
Percent of large fish	High	13.266	-0.1005	0.001	Yes	17
	Medium	9.892	-0.0380	0.231	Yes	17
	Low	10.229	-0.0114	0.819	Yes	17
Average fish weight	High	166.791	-0.9606	0.000	Yes	17
	Medium	152.467	-0.7084	0.009	Yes	17
	Low	156.236	-0.4754	0.219	No	18
von Bertalanffy Length _{infinity}	High	55.217	-0.1594	0.000	No	19
	Medium	56.211	-0.1873	0.000	No	19
	Low	52.955	-0.0547	0.238	No	19
Age at maturity	High	2.465	-0.0037	0.001	Yes	18
	Medium	2.398	-0.0019	0.023	Yes	18
	Low	2.373	-0.0007	0.364	No	19
Length at maturity	High	27.307	-0.0510	0.000	Yes	18
	Medium	28.253	-0.0610	0.000	No	19
	Low	26.669	-0.0154	0.460	No	19
von Bertalanffy growth	High	0.214	+0.0018	0.000	Yes	18
	Medium	0.227	+0.0017	0.000	No	19
	Low	0.230	+0.0008	0.000	Yes	18
Number of species	High	45.268	-0.2042	0.000	No	19
	Medium	37.389	-0.0396	0.265	Yes	18
	Low	28.687	+0.0319	0.310	Yes	18
Margaleff's species richness	High	4.380	-0.0215	0.000	No	19
	Medium	3.699	-0.0102	0.005	Yes	18
	Low	3.210	-0.0066	0.046	Yes	18
Pielou's species evenness	High	0.572	-0.0018	0.000	Yes	18
	Medium	0.540	-0.0018	0.000	Yes	18
	Low	0.489	-0.0006	0.299	No	19
Hills N1 species diversity	High	8.143	-0.0476	0.000	No	19
	Medium	6.739	-0.0331	0.000	Yes	18
	Low	5.129	-0.0072	0.393	No	19
Hill's N2 species diversity	High	5.809	-0.0300	0.000	No	19
	Medium	4.607	-0.0182	0.004	Yes	18
	Low	3.428	+0.0008	0.916	No	19
Nitrogen stable isotope ratio	High	14.181	+0.0051	0.193	No	18
	Medium	14.217	+0.0047	0.078	No	18
	Low	14.517	-0.0008	0.676	No	18

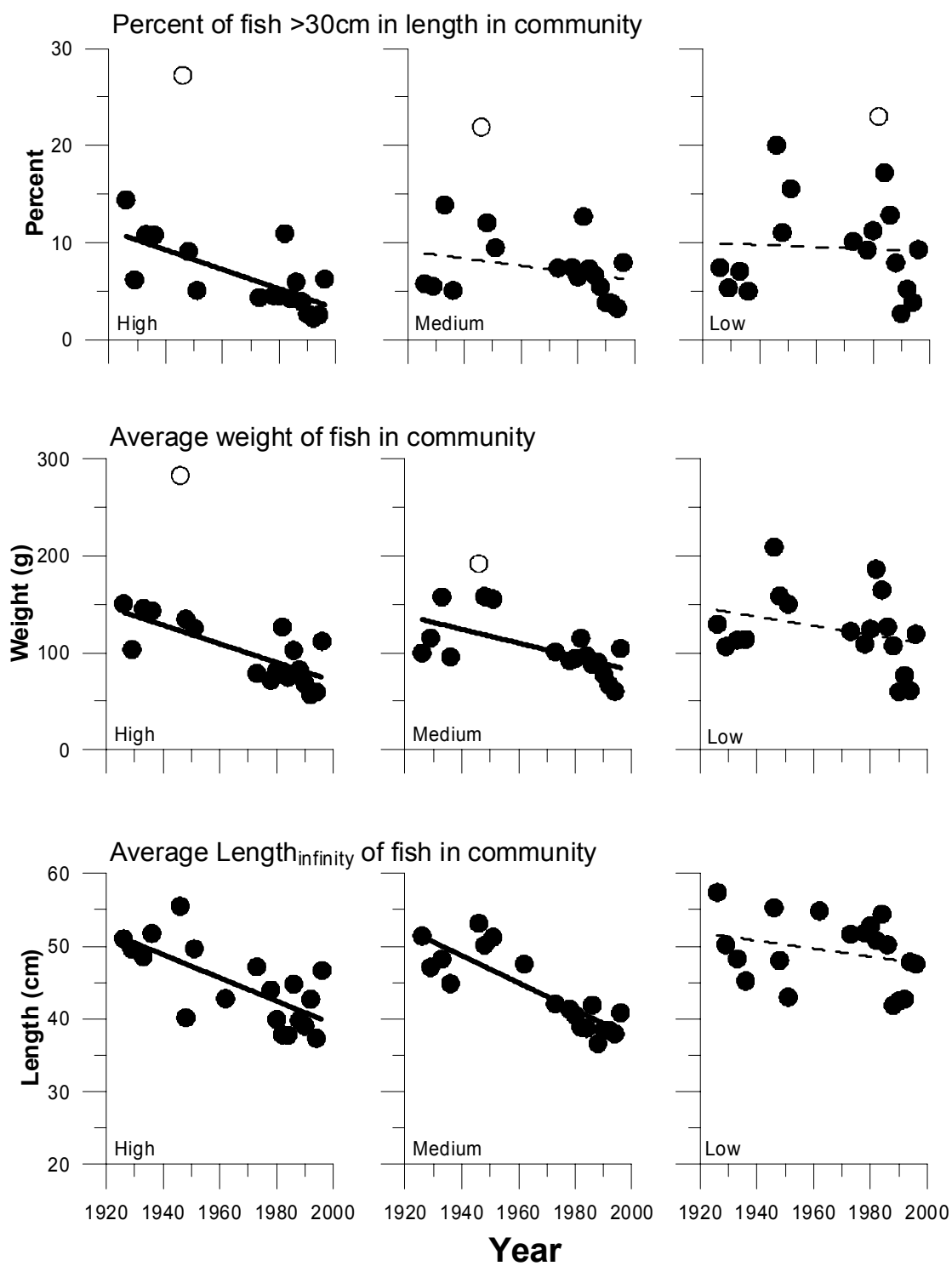


Figure 2.4.2.4. Long-term temporal trends in three fish community fish size metrics in three fishing effort level treatments.

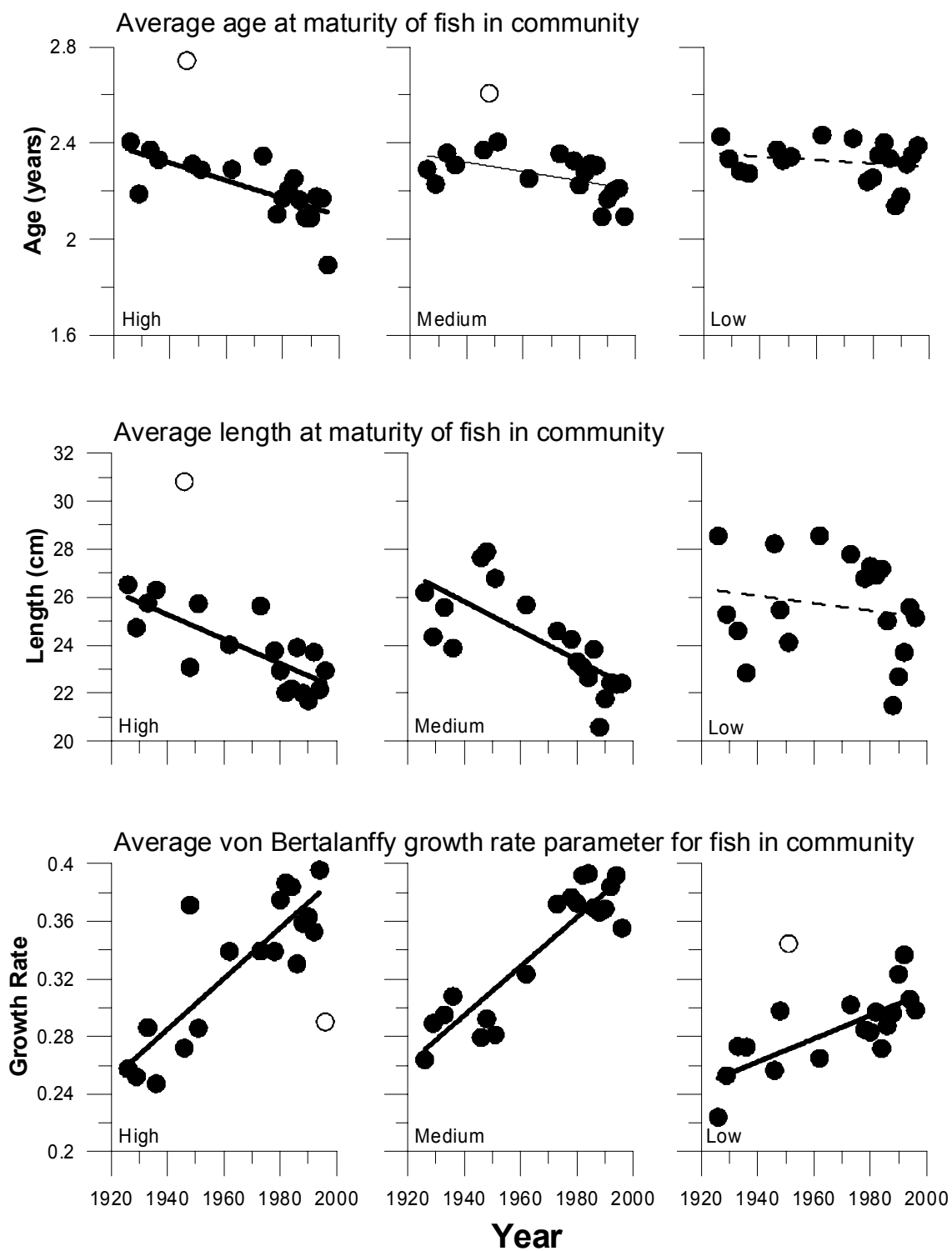


Figure 2.4.2.5. Long-term temporal trends in three fish community life history characteristics metrics in three fishing effort level treatments.

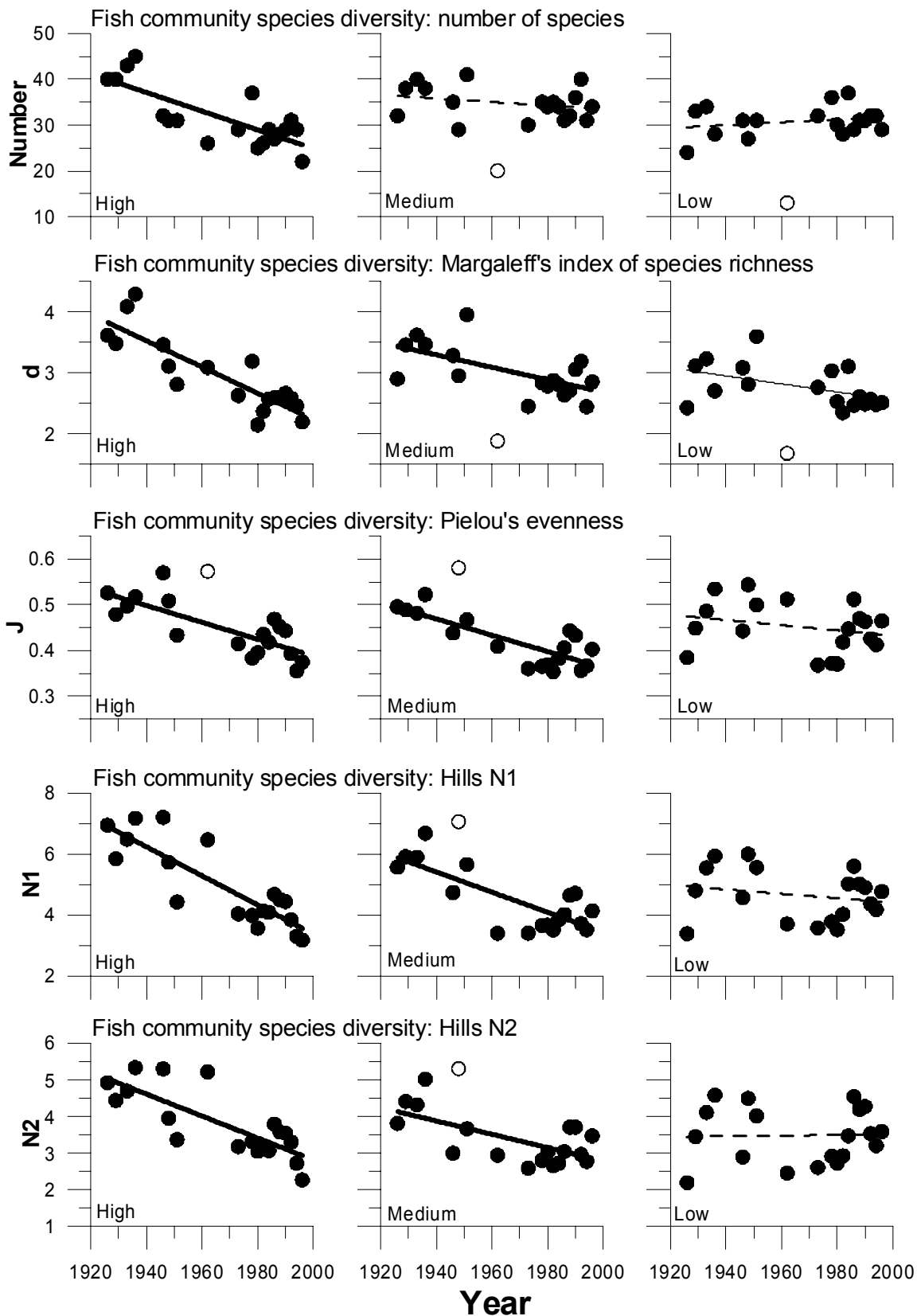


Figure 2.4.2.6. Long-term temporal trends in five fish community species richness and diversity metrics in three fishing effort level treatments.

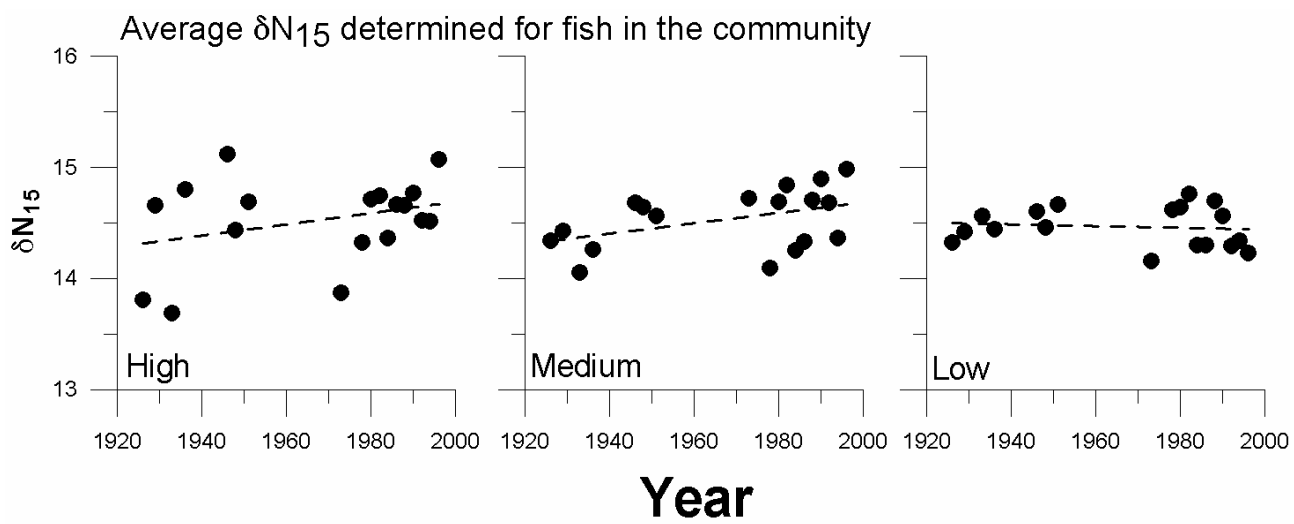


Figure 2.4.2.7. Long-term temporal trends in a fish community trophic structure metric in three fishing effort level treatments.

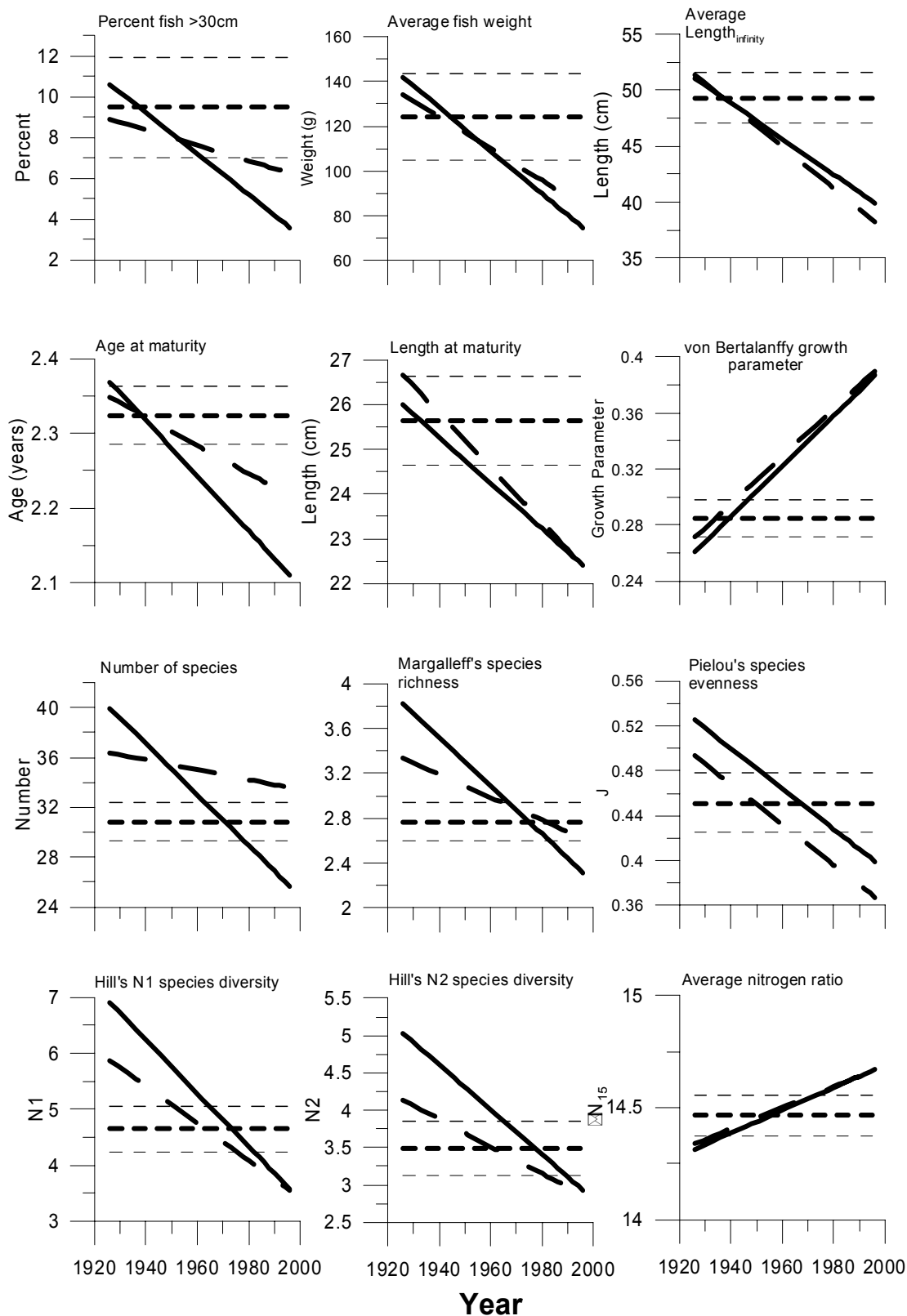


Figure 2.4.2.8. Long-term temporal trend-lines for the twelve fish metrics observed in the high and medium effort treatments compared with references levels determined from the mean and 95% confidence limits of the data in the low effort treatment.

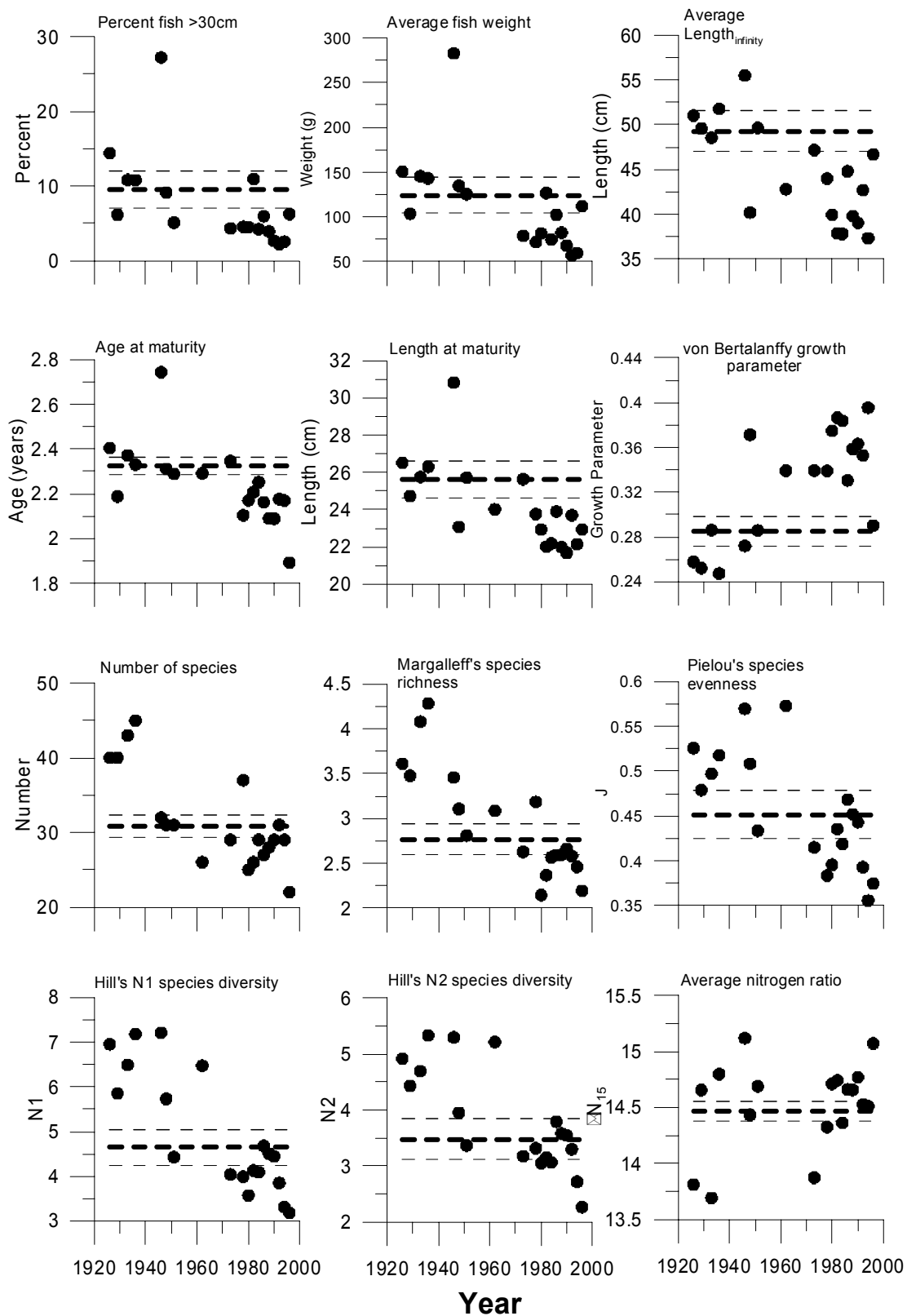


Figure 2.4.2.9. Long-term data for the twelve fish metrics in the high effort treatment rectangles plotted against references levels determined from the mean and 95% confidence limits of the data in the low effort treatment.

2.4.2.4 Discussion

The analyses presented here provide strong evidence that 11 of the 12 community metrics have responded strongly to variation in fishing effort. Thus where they were previously deemed to be weak by the application of the ICES criteria (Table 2.4.2.1) in respect of criteria b, c, and e, these shortcomings have now been redressed. The data on which the metrics are based, numbers and length of fish captured in trawl surveys, are easily and accurately measured by trained technicians on board research vessels. Such groundfish surveys are regularly carried out, and this has been the case for many decades in the North Sea and in other OSPAR areas. Thus these metrics should all score highly against criteria d, f, and g. Some of the metrics, e.g., fish size, are easily understood by non-scientists, although others are less transparent. It will remain the job of scientists to ensure that managers understand the messages being conveyed by more technical metrics. The theoretical linkages between the behaviour of some of the metrics examined here and variation in fishing activity is relatively well understood, for example the relationship between variation in fishing mortality and change in the size composition in fish communities. For other metrics the relationships are less obvious. Huston's (1994) dynamic equilibrium model for example suggests that species diversity may respond both positively and negatively to changes in disturbance. Managers need to know about changes in fish community species diversity in order to respond to policy drivers such as the CBD and Annex V of OSPAR. If at present marine science is incapable of explaining precisely the relationships between species richness and diversity metrics and variation in fishing activity, this does not mean that these metrics should be abandoned. Instead research activity should be directed towards improving our knowledge of the theoretical basis for these metrics that managers require.

The Nitrogen ratio showed no significant difference between the three effort treatments in contemporary times, and no long-term trends were detected in either of the high or medium treatments. Given the strong suspicions that fishing has affected the trophic structure of fish communities, these results are slightly difficult to accept, and can be interpreted in a number of ways in relation to Rice's (2003) signal theory. Either fishing has not affected fish community trophic structure, in which case this result is a "true negative". Alternatively, fishing has affected community trophic structure, but that this metric is either insensitive to the fishing effect, or is perhaps not a good indicator of trophic structure. In either case, the result should be considered as a "miss". If there are serious doubts about this being a "true negative", then this result clearly highlights the need for more research, both to examine trends in the trophic structure of the North Sea fish community and to identify a reliable indicator of the trophic functioning of fish communities.

The significant log-term temporal trends in Margaleff's species richness index and the von Bertalanffy growth parameter in the low effort treatments perhaps indicates an environmental effect on these parameters. Warming water temperatures may for example give species with higher growth rates a selective advantage. With increased productivity, species richness may have declined. A fish community increasingly consisting of faster growing individuals may be more susceptible to single species "outbreaks", and therefore increasingly likely to become dominance orientated. Whatever the reason for these two significant trends, they call into question the wisdom of simply using the mean and 95% confidence limits of the low effort treatment as "reference levels" for these two metrics. EcoQOs set against such "reference levels" for these metrics may simply be unattainable given changes in the factor(s) responsible for the significant trends in areas where fishing effort has been so low that fishing is unlikely to have been the cause. In such circumstances the regression relationship for the low effort treatment, along with its 95% confidence limits might be used instead to set "reference levels more appropriate for the current times.

For the other 9 metrics examined, the analysis presented here would seem to illustrate a feasible method for assessing realistic "reference levels" on which to base EcoQOs for different attributes of the fish community. Interpretation of the suite of indices applied here provides a considerable amount of information regarding changes in the fish community itself, and about the potential role of the fish community within the broader North Sea ecosystem. The data would appear to suggest that fishing has caused a reduction in the size and weight of fish in the assemblage, and that the current demersal fish community is now dominated by species whose ultimate body length is shorter than in the past. Other life history characteristics have also changed. The assemblage is now more dominated by species whose growth rates are faster, and which mature at an earlier age and smaller body-length. This suggests that the fish assemblage is more productive and potentially more capable of recycling resources at faster rates. If one assumes that the lack of change in the Nitrogen Ratio is in fact a "true negative", and that the trophic structure of the assemblage has remained unchanged, then this also has important implications. It implies that the larger, less productive, and presumably mostly *piscivorous* species, which have been lost from the community have been replaced by smaller, faster growing fish that are also *piscivorous*. Trophic structure may be ecologically constrained and, so far, robust to the factors causing change in fish communities. If recycling rates have increased, but the trophic structure has remained unchanged, then this has important implications for ecosystem dynamics, and underlines the need for research into this topic.

Along with all these other changes in the North Sea fish community, fishing activity does appear to have caused a reduction in the species richness and species diversity of the assemblage. There are sound theoretical reasons to be concerned about the implications for ecosystem function that such changes might hold. More importantly, however,

these are changes that managers are obliged to respond to given the obligations made to, for example, the CBD and OSPAR. These commitments require managers not only to conserve biodiversity, but also to restore biodiversity in situations where declines in biodiversity have been detected. The data presented here suggest that managers adopting an ecosystem approach to management have an immediate task to address.

Elsewhere in this report (see Sections 6 and 7), and in the reports of other ICES working groups, the importance of recognising that groundfish survey data provide a “gear-biased” view of the fish communities they sample is highlighted. The “picture” obtained is far from being a true view of the “real world”, rather they are the perception we have, based on the catches of the sampling gear used. Thus trends in the metrics presented here are the trends as detected using an Aberdeen 48 foot demersal trawl. This does not invalidate the conclusions drawn. However, it is important to apply these community metrics to a variety of different surveys in order to determine how universal these signals are. Different groundfish surveys would in all likelihood pick up the same trends, but the actual metric values may very well differ between the different data sets derived from different trawl gears. This becomes an important issue when it comes to setting EcoQOs. The data set analysed here is the longest lasting continuous groundfish survey carried out in the North Sea, long enough apparently to have detected the deviation point away from the non-anthropogenically affected state for most of the metrics examined here. This data set is therefore ideal for the identification of “reference level” against which EcoQOs might be set. However, this survey was discontinued in 1997, and is no longer available as a tool to monitor progress towards such EcoQOs. Comparative studies between different trawl surveys are essential as progress towards EcoQOs set on the basis of the SAGFS will have to be monitored using data gathered using a different sampling gear.

2.4.3 Comparison and utility of different size-based metrics of fish communities for detecting fishery impacts

The purpose of the working paper “Comparison and utility of different size-based metrics of fish communities for detecting fishery impacts” by Duplisea and Castonguay (submitted) was to define the utility of each type of size spectrum analysis for revealing information about the total fish community and impacts of exploitation.

2.4.3.1 Introduction

Given the multitude of methods for size spectra construction (see Section 2.2.2), multiple statistical model fittings and underlying theory, size spectra work can be confusing and work has been done to relate and clarify the methods (Han and Straškraba 1998). It is however important to determine which analyses might be useful for particular questions or under which conditions. In this working paper we examined size spectra patterns using various methods for six very high quality fish survey data sets. It is our purpose to define the utility of each type of size spectrum analysis for revealing information about the total fish community and impacts of exploitation. It is hoped that such an analysis will help define how size spectra might be used in an ecosystem approach to fisheries management.

2.4.3.2 Methods

Six different regions (Figure 2.4.3.1) were studied by examining data from four separate sources: (1) Atlantic Canadian groundfish summer survey for the Scotian Shelf (2) the Canadian February trawl survey on Georges Bank (3) Northern Gulf of Saint Lawrence groundfish and shrimp summer survey (4) the English groundfish summer survey for the North Sea. From survey 1 we obtained data for the Cabot Strait (4Vn), the eastern (4VsW) and western (4X) Scotian Shelf from survey 2 we obtained data for Georges Bank (5Z) and from surveys 3 and 4 we obtained data for the northern Gulf of Saint Lawrence (4RS) and North Sea (IVa,b,c), respectively.

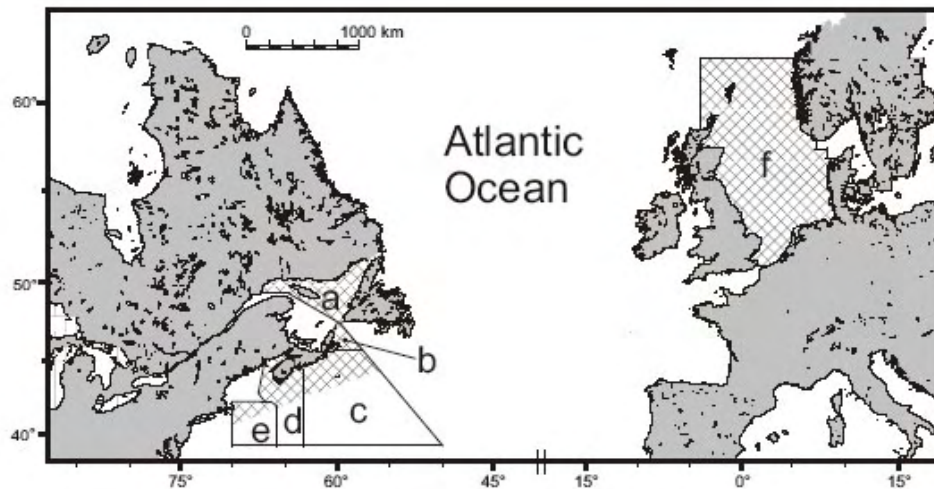


Figure 2.4.3.1. Map of the different systems studied with statistical management areas demarcated: (a) 4RS, N. Gulf of St. Lawrence (b) 4Vn, Cabot Strait (c) 4VsW, E. Scotian Shelf (d) 4X, W. Scotian Shelf (e) 5Z, Georges Bank (f) IVa,b,c, North Sea. Surveys represent continental shelf and shelf slope areas, which are roughly shown as the hatched areas inside the larger statistical areas. Systems (a)-(e) are managed under the Northwest Atlantic Fisheries Organisation (NAFO) and system (f) is managed under the auspices of ICES. Management area coding refers to the schemes of the respective organisations.

Each of these survey types has different protocols for fishing and gear and different modes of data storage; hence they were treated separately and are not directly comparable in all respects. They are, however, comparable in all analyses conducted here. The Canadian survey are represented here under the statistical areas, and abbreviation conventions of NAFO (www.nafo.ca) and the English survey is represented under the statistical areas and abbreviation conventions of ICES (www.ices.dk).

The Canadian surveys utilise a stratified random sampling design (Doubleday 1981). Generally a 3.2 km standard tow of a 12 m (width at mouth) otter trawl is conducted at randomly chosen stations within each defined depth stratum (total number of tows, range). For each tow, all the fish are weighed and a subsample (200 individuals per species) is taken for computing length-frequency distributions. These length frequency distributions were the basis of the data used here. They were with either used directly as length-frequency distributions or weights and biomasses were calculated from species-specific length-weight regressions. The data from each standard tow was averaged within strata and then a weighted average of strata tows was computed with weights equal to the surface areas of each stratum. This is a standardised procedure known as STRAP (Smith and Somerton 1981) and provides a representative average over the area of interest. For the eastern and western Scotian Shelf and Cabot Strait, the survey time series runs from 1970 to 2001 while the Georges Bank survey commenced in 1986. A standard tow in this survey is 1.75 nautical miles (1 nautical mile = 1.13 km).

The survey in the northern Gulf of St. Lawrence is similar to the other Canadian surveys. This survey commenced in 1978 but the vessel changed in 1984 and then in 1990 the vessel and gear changed from VIIa (western 2 a) trawl to a URI (University of Rhode Island) shrimp trawl which is more suitable for sampling the shrimp populations in the northern Gulf of St. Lawrence. This survey has remained constant since 1990 and runs to 2002 so only this part of the survey was used. A standard haul is 1.75 nautical miles (1 nautical mile = 1.13 km).

The English groundfish survey of the North Sea is conducted by performing standard tows of ½ to 1 hour duration at fixed stations each year. Between 75 and 100 stations are towed each year during daytime hours only. For these stations all species are weighed and subsamples of individuals of each species are weighted. Length-frequency or weight-biomass distributions were calculated for each station. Stations were then averaged to obtain an average spectrum for the North Sea. The North Sea survey data used here, run from 1977 to 1999. A standard tow in this survey runs for 1 hour at 4 knots.

In each system we included fish species and squids (e.g., *Loligo* and *Illex*). Data consisted mostly of teleosts but also a large elasmobranch component was represented in all the systems. Squids usually constituted only a small portion of biomass in any tow. For each system we constructed two types of multispecies body-size abundance distribution:

- 1) a length-frequency distribution, where 5 cm body length categories were used and abundance within each 5 cm length bin was summed and logged (Figure 2.4.3.2a).
- 2) log2 biomass at log2 body weight distributions, commonly known as biomass spectra (Figure 2.4.3.2b).

For both analyses, data were prefiltered to include fish only of lengths between 15 and 150 cm. Linear regressions of log abundance versus length were computed for each year in each region (Figure 2.4.3.2a). From the biomass spectra three metrics were derived and examined for temporal changes: the curvature of a quadratic fitted to the spectrum, the body size at the vertex of this fitted parabola and, the standardized biomass at the vertex of the fitted parabola (Figure 2.4.3.2b). Temporal trends in these fitted size spectra parameters are examined in this paper. Data points in these plots are represented as the regression r^2 value (i.e., a measure of the quality of the regression fit) for a year.

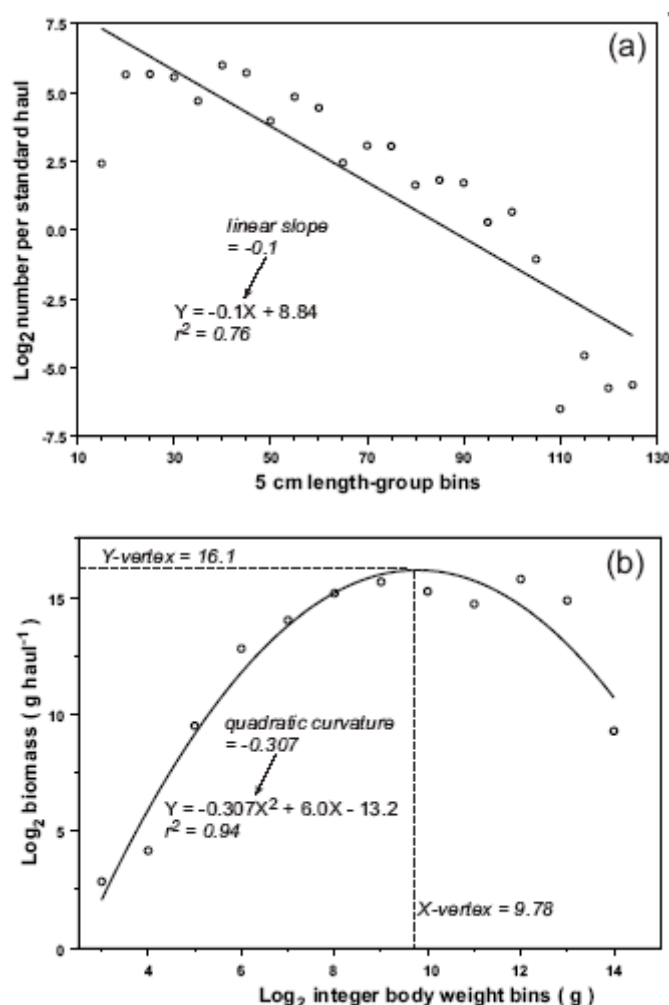


Figure 2.4.3.2. Demonstration plots of (a) a straight line slope fitted to log number versus length size spectra (b) a quadratic fitted to log biomass versus log body weight size spectra. Least square fitted size spectra parameters (linear slope, quadratic curvature, X-vertex, Y-vertex and r^2) analyzed in this study are shown on the plots. Data shown are for Georges Bank survey data from 1987.

2.4.3.3 Results

Slopes of length frequency plots for all systems but the N. Gulf of St. Lawrence (the shortest of the time series) steadily declined since the survey commenced (Figure 2.4.3.3). Georges Bank and the eastern Scotian Shelf time series show the most marked decline. Georges Bank in the earlier years (1986) and the Cabot Strait in the 1970 had the shallowest

slopes indicating that large fish were relatively more abundance in those systems then. The northern Gulf of St. Lawrence data shows a variable slope throughout the series yet an overall decreasing trend after an increase in the early 1990s.

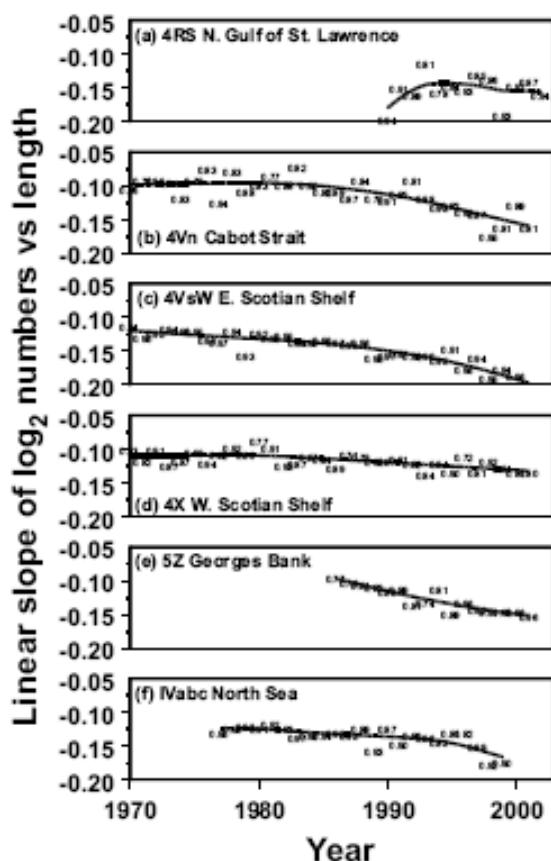


Figure 2.4.3.3. Linear slope of a regression fitted to \log_2 numbers per standard haul versus body length over time from survey data. Solid lines represent spline smoothers through points. Data points are depicted by the r -squared value of the linear regression fit of the slope to data in that year.

The curvature of the biomass spectrum (Figure 2.4.3.4) for Georges Bank decreased (the parabola was tighter) between the mid 1980s and the early 1990s but levelled off after this time. The other systems had essentially no sustained changes in curvature over the time period. None of the parabola fits showed any temporal trend in any system but the northern Gulf of St. Lawrence has more poor parabola fits over the time series and the most variable time trend relative to the other systems (Figure 2.4.3.4). There is some indication of a periodicity in curvature for the Cabot Strait (8 year period, $r=0.28$) however this is not quite significant at the 95% level. A 3 year period is present in curvature for the western Scotian Shelf ($r=0.35$, $p=0.05$) (Figure 2.4.3.4).

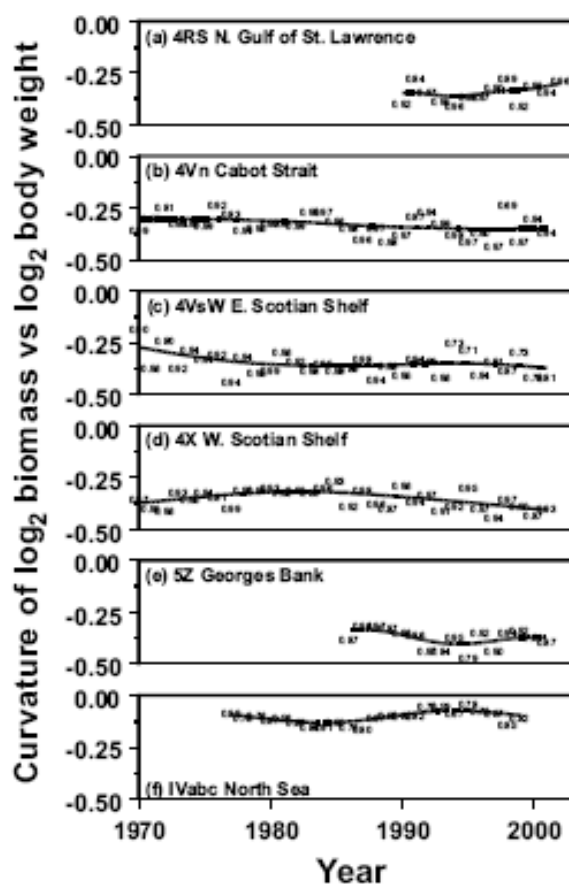


Figure 2.4.3.4. Quadratic curvature of regression fitted to \log_2 biomass per standard haul versus \log_2 body weight over time from survey data. Solid lines represent spline smoothers through points. Data points are depicted by the r-squared value of the regression fitted of the quadratic to data in that year.

Body weight at the vertex of a parabola fitted to the biomass spectrum is roughly a measure of the modal or average body weight in the fish community. In the northern Gulf of St. Lawrence, body size at the vertex increased from 1990 until 2000 (270 g to 670 g) but decreased by 50% in the two following years (Figure 2.4.3.5). For all the other systems, body size at the vertex was relatively constant until 1980 after which it decreased. This trend was most apparent for the Scotian Shelf and Cabot Strait, where the longest time series exist.

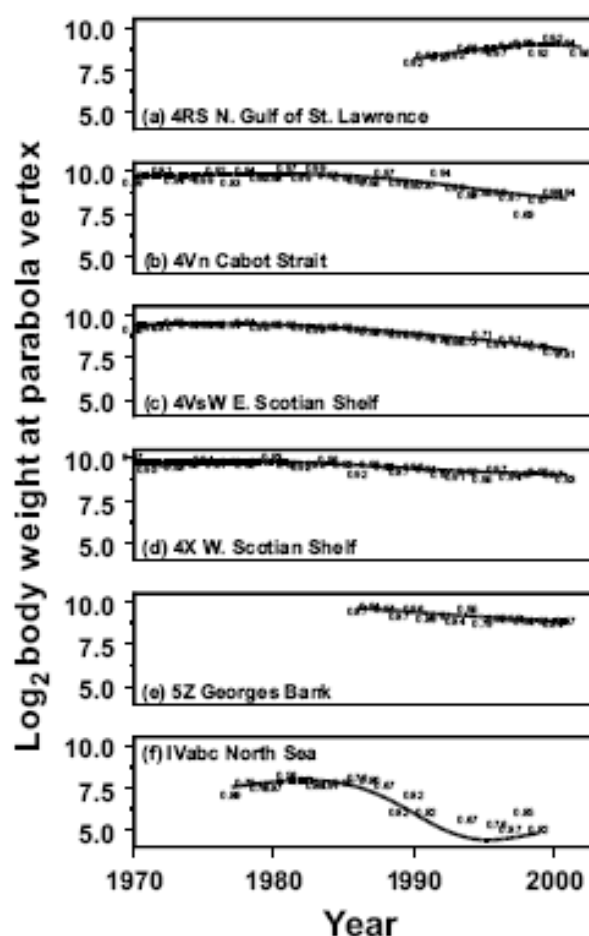


Figure 2.4.3.5. Body-size at the vertex of a quadratic fitted to log₂ biomass per standard haul versus log₂ body weight over time from survey data. Solid lines represent spline smoothers. Data points are depicted by the r-squared value of the regression fitted of the quadratic to data in that year.

The decrease in body size at the vertex was most dramatic in the North Sea where body size decreased by an order of magnitude in the mid to late 1980s. The western Scotian Shelf was the system with the largest vertex body size on average, though not appreciably greater than in other western Atlantic systems. The North Sea, on the other hand, had a modal body size considerably lower than the other systems, i.e., biomass is more concentrated in small fish in the North Sea. This was particularly marked in the 1990s where large biomass of small fish made the size spectrum bimodal and fitted parabola had to accommodate both of these modes, hence the poorer fits in these years for the North Sea.

Biomass at the peak of fitted parabolas (Figure 2.4.3.6) for the Cabot Strait and eastern Scotian Shelf decreased from the late 1980s onward whilst in the other systems the value remained fairly constant. In the North Sea a low in biomass occurred for several years in the late 1980s and early 1990s. No clear trends in the quality of fits (represented by r^2 values as data points in Figures 2.4.3.3–2.4.3.6). Generally quality of regression fit of straight lines and quadratics was quite high (r^2 usually >0.80). In all systems but the North Sea, a quadratic fitted to biomass vs body weight size spectra was of about the same quality of fit as a straight line fitted to numbers versus body length size spectra. The best overall regression fits were for spectra from the eastern Scotian Shelf which were well fitted with both the quadratics and straight lines and did not vary considerably over the time series.

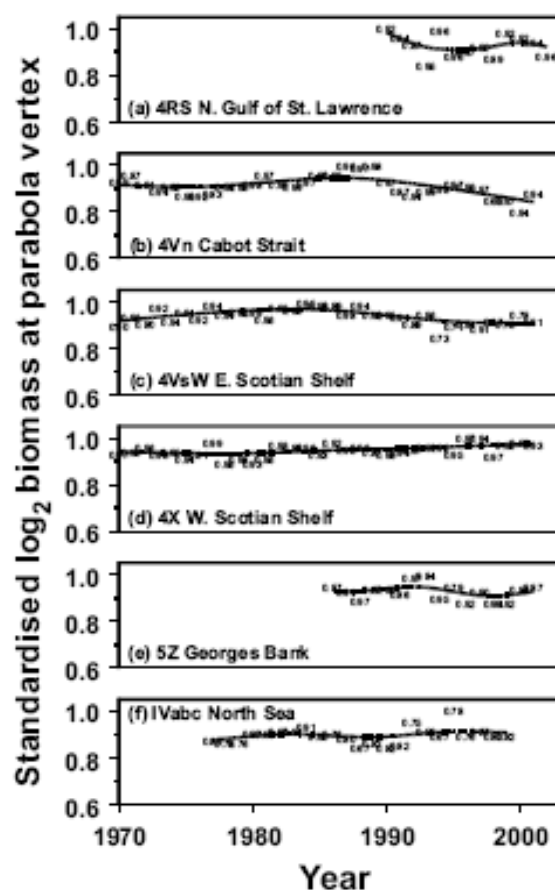


Figure 2.4.3.6. Standardised Log2 biomass at the vertex of a quadratic fitted to log2 biomass per standard haul versus log2 body weight over time from survey data. Values are standardised by the largest value in the series for each system. Solid lines represent spline smoothers through points. Data points are depicted by the r-squared value of the regression fitted of the quadratic to data in that year.

Total system standardised biomass shows the variability in biomass over time (Figure 2.4.3.7) in some systems this varied more than others while in other systems trends were present. The N. Gulf of St. Lawrence and North Sea were quite variable while the Cabot Strait and Eastern Scotian Shelf total biomass decreased over time. The Western Scotian Shelf was the only system which showed increasing biomass over the survey time series.

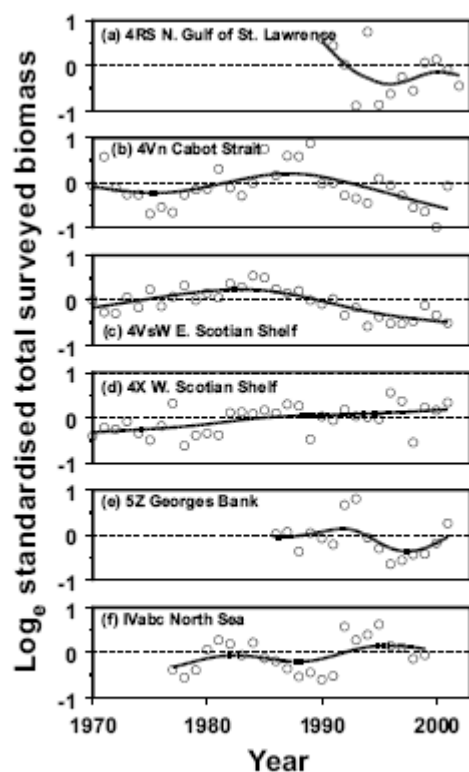


Figure 2.4.3.7. Logged standardised total biomass per standard haul from survey data. Values are standardised by the mean value of the series for each system. Solid lines represent spline smoothers through points. The dashed line represents the mean total surveyed biomass.

2.4.3.4 Discussion

Various size spectrum measures provide complementary information on the state of systems. For example, the linear size spectrum slope is indicative primarily of the decline in abundance of the larger fish in a system, which has declined in all of the studied systems. The body weight at the vertex of a quadratic fitted to biomass spectra, however, is less sensitive to the abundance of large fish in the system but is an indicator of the body size of fish where the biomass peaks hence is more equivalent to a mode. These are different pieces of information with different sensitivities to perturbation. The linear slope is sensitive to abundance of large and small fish rather than to body size where the bulk of the biomass is located. Consequently, linear slopes are relatively sensitive to gear and vessel changes that affect the catchability of fish at the endpoints of the sampled size range. Modes are inherently more robust measures of populations hence changes in their values are indicative of large differences in underlying processes. Modes better describe what is happening in the middle sizes rather than by the extreme sizes of the fish size distribution hence are more robust to changes in sampling methods which most often affects the end points of sampled body size ranges.

The linear size spectrum slope clearly shows the decline in large fish abundance over time in most systems. This is particularly apparent for Georges Bank and the eastern Scotian Shelf both of which have shown dramatic declines in abundance of the large gadoid component of their communities. The western Scotian Shelf in contrast has not shown nearly the same degree of fishery collapse and decline in large fish abundance which is reflected in the size spectrum linear slope which concurs with the known patterns in these systems.

Size spectrum slope may be a good indicator of increasing fisheries pressure within a system provided that high quality survey data are available. Size spectrum slope though is quite sensitive to changes in survey methodology especially in factors that will affect catchability of large fish. Unfortunately, catchability of large fish is one of the most easily influenced factors in surveys that can be caused by gear changes and/or changes in trawlable habitat available to certain gear type. For this reason, it is tenuous to compare the slope values between systems with different kinds of surveys or even to compare time trends in slopes between systems if there have been changes in survey methodology over the time

period. This fact limits the utility of size spectrum linear slope largely to systems in developed countries where high quality long-term survey data exist. The great advantage of slope is that it is so easily interpreted.

The curvature of a quadratic fitted to biomass size spectra is less easily interpretable than the linear size spectrum slope. Previous work on the Scotian Shelf as a whole (4VX), however, showed that size spectrum curvature reflected two stanzas in fisheries management surrounding the implementation of the 200 mile limit international law of the sea convention in 1978 (Duplisea and Kerr 1995). Analysis of sub-regions of the Scotian Shelf here show no clear indications of this. Though the size spectrum curvature itself may not present clear signals, other parameters derived from quadratic fits do appear more useful:

The X-vertex of fitted parabolas is one of the most robust and easily interpretable size spectrum parameters derived from quadratic fits. The signal is not noisy given gear changes etc. and it is interpretable because it reflects the modal body size of fish biomass in the system. It clearly corroborates the decreasing linear slope for Georges Bank size spectra and indicates a decreasing trend for all Scotian Shelf systems beginning in about 1985. This is indicative of our knowledge of the decline of fisheries on the Scotian Shelf over this time and also corroborates the decline in fishing success even after fisheries conservation measures were enacted in the mid 1990s. The increase in the vertex body size in the Northern Gulf of St. Lawrence reflects the 1993 moratorium on cod fishing which allowed the small cod to grow older and larger. The decreasing body size after 2000 likely reflects the reopening of the cod fishery in 1997 and the reoccurrence of worsening conditions (CSAS 2003).

The drastic decline in the modal body size (X-vertex) in the North Sea in the late 1980s reflects the recovery of herring populations along with a decrease in groundfish abundance over that period. The size spectrum for the North Sea in these years looks bimodal reflecting the abundance of the smaller pelagics and subsequently poorer fits of the parabola to data in those years. The subsequent increase in X-vertex and improving fit likely reflects the maturation of some good recruitment years for groundfish, most notably the 1996 age class of cod.

Curvature of quadratic fits are useful in total system studies which examine several trophic groups such as phytoplankton zooplankton and fish (Sprules and Goyke 1994). Curvatures in this context are known to be repeated between trophic groups. Curvatures have been theoretically derived from parameters describing trophic processes such as prey/predator size ratio and predator foraging efficiency (Thiebaut and Dickie 1993).

Hence curvature will reflect changes in these processes in a system and theoretically throughout all the trophic groups. Therefore it is not surprising that curvature is relatively unchanged for a single trophic group (fish) within systems and also indicates why this might be more useful for intersystem comparison as an indication of the body size range over which biomass is distributed. Therefore, curvature may show changes with species diversity and foodweb complexity and perhaps a latitudinal gradient concomitant with these other changes.

Quadratic curvature will reflect two processes in an exploited system which tend to counteract each other. That is, the right arm of the fitted quadratic will tend to track the decline in abundance of large fish. The left arm, however, will tend to follow in increasing abundance of small fish which are released from predatory control (Duplisea and Bravington 1999). The first process will tend to make the quadratic curvature more negative while the latter will make it more positive. The net result is a relatively insensitive quadratic curvature not overly useful for documenting within system trends in community response to fishing.

The biomass at the vertex of parabolas fitted to biomass size spectra (Y-vertex) is mostly a reflection of the total biomass in system and is analogous to the intercept of a straight line fitted to BSS (Bianchi *et al.* 2000). Often it seems that this value increases initially with exploitation and then decreases after prolonged exploitation. Such a pattern might be explained by predation release on prey followed by stock collapse. That is as fisheries begin to reduce the abundance of large predators in the system, small fish (prey) biomass can increase. This results in an increasing Y-vertex while X-vertex decreases. However, with continued switching in multispecies fisheries eventually populations of small fish also decrease in abundance indicating a more systematic collapse.

We plotted Y-vertex vs. X-vertex for these systems and for the eastern Scotian Shelf (the most marked pattern) we found this pattern indicating a decline towards the origin in the most recent years (Figure 2.4.3.8). Though this is a compelling indicator it must be interpreted cautiously as it is not universal in all the systems here but the pattern accords with the devastating decline in groundfish stocks on the eastern Scotian Shelf.

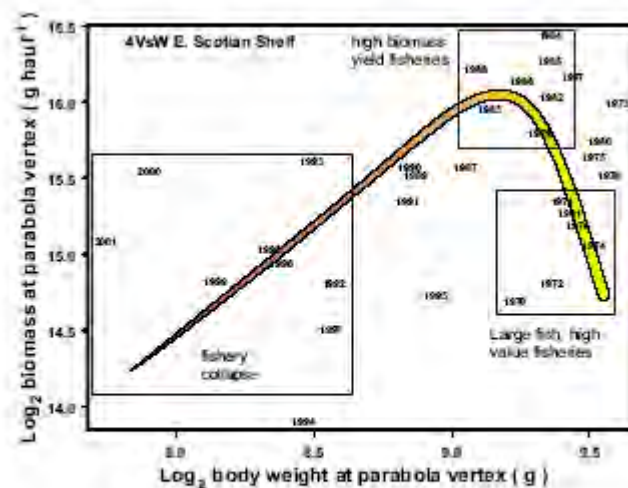


Figure 2.4.3.8. Pattern of fish community response to long term unsustainable fishing as shown by plot vertex coordinates of fitted parabolas against each other for all survey years using Eastern Scotian Shelf survey data. Boxes drawn around groups of years is not statistical but illustrative of significant periods relating to fisheries status and management. The stylised curve represents the trajectory going from productive fisheries system to collapsed system incapable of supporting much fishing.

The pattern depicted in Figure 2.4.3.8 represents the trajectory of a total community (by embodying characteristics of body size and biomass) in response to fisheries exploitation. An analogue for such a plot may be Holling's "lazy-8" (Holling 1992), that describes the cycles of building, destruction and regeneration in the in a cyclical pattern represented by a horizontal "8". The lazy 8 is a valuable heuristic model which helps interpret ecosystem changes in terms of natural changes in self-organising system like ecosystems. Though the exact shape of the Lazy-8 model should not be interpreted too literally or quantitatively, we modified Holling's diagram to accord with the size spectrum pattern in exploited fish communities and particularly with the pattern shown in Figure 2.4.3.8. The three phases of fish community state in Figure 2.4.3.8 are now depicted as part of a loop (Figure 2.4.3.9).

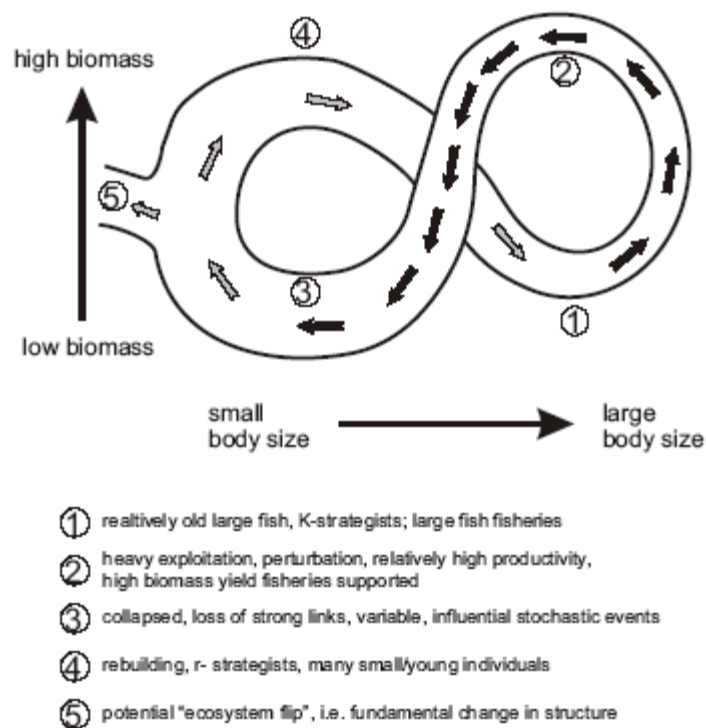


Figure 2.4.3.9. The “lazy-8” from Holling (1992) modified in order to represent the situation for an exploited temperate/boreal marine fish community as depicted by size-spectrum parameters (see Figure 2.4.3.8). The closeness of the arrows represents the rate of change in state; the black arrows represent the states observed (see Figure 2.4.3.8) while the grey arrows are hypothetical and have not yet been observed. The width of the path in stages 3 and 4 represents the large variability of the size structure then and the potential to occupy a much larger range of possibilities in state than at stages 1 and 2.

If such a loop actually exists, it is likely that recovery of the community (if it is allowed to occur) will not follow an exact reversal of decline but must follow a course of rebuilding biomass in small sizes succeeding into large long-lived individuals as were present before the 1960s. After collapse and into a rebuilding stage, there is greater potential for surprise in the system owing to weakly coupled energy flows and considerable temporal variation in the size structure of the community. It is at this stage that a fundamental change in state could potentially occur after which the previous community might never appear again, even if unperturbed. Given the high abundance of invertebrates (shrimp and snow crab) on the eastern Scotian Shelf and the failure of groundfish stocks to recover since a 1993 moratorium (Fu *et al.* 2001), it may be that this system has already followed an alternative pathway and may not recover to its previous state even in the absence of fishing.

2.4.3.5 Summary: utility of size spectrum parameters

- size spectrum linear slope is useful for looking at time trends within a system provided statistically rigorous consistent data sets are available. They generally reflect the abundance of large fish. They are of limited utility with inconsistent data set or with multiple gear changes because of the sensitivity to sampling of large fish. For the same reason they can rarely be used for direct parameter comparison between systems. They corroborate declines in fish sizes, fishing down the foodweb and oceanographically induced distributional changes in fish provided high quality data are available.
- size spectrum quadratic curvature is very robust to survey changes and differences, however they are relatively insensitive to fishery or oceanographic induced changes in the fish community. Curvature is therefore not very useful for within system comparison. However, because of the robustness of parabola fits and because they represent underlying general ecological structuring mechanisms other parameters derived from them are useful in other contexts:
- body size at the vertex of fitted parabolas (X-vertex) is analogous to a mode which reflect the most common (by biomass rather than abundance) body size in the system. X-vertex corroborates linear size spectrum slope in its

response to fishing indicating declines in body size with increasing fishing. This signal is however less noisy than size spectrum linear slope because it carries the same robust characteristics of the quadratic fits that curvature has. X-vertex is a valuable intersystem comparison parameter.

- biomass at the vertex of a fitted parabolas (Y-vertex) closely will track total system biomass values if no major changes are occurring in very small or large size fish. Y-vertex may increase with low exploitation levels usually owing to a small decrease in the body size of fish in the system to smaller individuals with higher production. A decrease in Y-vertex concomitant with X-vertex may indicate a more systemic collapse in fish stocks.
- size spectra may be able to track change in system state in response to fisheries exploitation consistent with Holling's four ecosystem function diagrams.

2.4.4 Statistical Power of size-based indicators

2.4.4.1 Introduction

Community metrics for ecosystem-based fishery management should be sensitive to fishing impacts and should respond rapidly to management action, such that managers can assess whether changes in fish communities are a desirable or undesirable response to management. It should be possible to estimate metrics with sufficient precision so that changes in the fish community can be detected on management time scales of a few years. As demonstrated by the case studies presented in this report most metrics are calculated from species-size-abundance data collected from fish stock monitoring surveys. Nicholson and Jennings (2004) showed that in order to obtain a 90% power of detecting trends, the number of years of monitoring required for several size-based indicators under the maximum and likely rate of change under existing management ranged from 12–30 to 30–75 years. Although the metrics based on current monitoring may provide good long-term indicators of changes in fish community structure, at short time scales statistical power is weak. The following analyses focused on the power to detect trends in size-based metrics and demonstrate that power may also depend on the choice of the range of size classes used in the analysis (Jennings and Dulvy, in press).

The following analyses focused on the power to detect trends in size-based metrics and demonstrate that power may depend on the choice of the range of size classes used in the analysis (Jennings and Dulvy, in press).

2.4.4.2 Methods

The power to detect trends in the direction of change in size metrics was calculated using data from the North Sea International Bottom Trawl Survey (IBTS). Community metrics were calculated from species-size-abundance data for 107 rectangles sampled in every year from 1982–2000, based on a standard tow duration of one hour. Mean values of the following metrics were then calculated among rectangles within years.

Mean body mass was calculated as

$$\bar{M} = \sum M / N \quad (4)$$

where M is the body mass of an individual. N is the total number of individuals.

The *slopes of biomass size spectra* were calculated by assigning individuals to \log_2 body mass classes, and cumulative biomass by body mass class was calculated. The linear relationship between \log_{10} body mass midpoint of \log_2 body mass class (x) and \log_{10} biomass (y) was described as $y = bx + a$, where b is the slope of the size spectrum.

Mean maximum mass was calculated as

$$\overline{M_{\max}} = \sum (M_{\max_j} N_j) / N \quad (5)$$

where M_{\max} is the maximum body mass attained by species j and N_j is the number of individuals of species j . Only mature fishes, defined as those longer than the length at 50% maturity (L_{mat}), were included in this metric so that it reflected the composition of life history traits in the spawning component of the fish community. Further details of the IBTS data and metric calculations are provided in (Jennings *et al.* 2002; Nicholson and Jennings 2004).

To identify the size window that optimised power, and hence the tradeoff between trend and variance, we calculated the power to detect future trends in the community metrics for fishes in classes from $x=4096g$, where x varied from 16 to 256g. As well as the magnitude and pattern of trend, sampling scheme and significance level of the test, power depends on the magnitude of the residual variance, ψ^2 (Nicholson and Fryer 1992). Variance was estimated using the difference-based variance estimation method recommended by Gasser *et al.* (1986). This has the form

$$\hat{\psi}^2 = \frac{2}{3(T-2)} \sum_{t=3}^T (0.5y_{t-2} - y_{t-1} + 0.5y_t)^2$$

for a time series of equally spaced observations y_t , where $t = 1 \dots T$. The estimated variance was then used to derive the power to detect a reversal in trends of community metrics.

2.4.4.3 Results

The power calculations show that metrics calculated from intermediate body mass classes provide the greatest power to detect trends (Figure 2.4.4.1). For mean mass, the power to detect future trends is maximised when $x=32g$, while for the slope of the size spectrum and mean maximum mass, power is maximised at $x=64g$. However, an appropriate level of power to detect change in metrics will not be achieved for a number of years.

2.4.4.4 Conclusions

The power calculations suggest that the ability to detect a reversal of trend in size metric is low. However, this analysis assumes that the estimate of variance from retrospective analysis will also apply in future years. This assumption will be violated when fishing mortality and the environment change. Thus, when fishing mortality rises, the age structure of the population is likely to be truncated and individual recruitment events will lead to greater relative changes in abundance. Conversely, when fishing mortality falls and abundance rises, interannual variance in abundance is likely to fall because the effects of annual variations in recruitment will be buffered by the increasingly extended age structure of the population and density dependence. Consequently, true power will be lower than predicted when abundance is falling and higher than predicted when abundance is rising (Jennings and Dulvy submitted).

Also this test was particularly stringent in that power was calculated to detect a *reversal* of trends and thus a move toward recovery or restoration of size structure, as measured by these size metrics. In reality a more preferable or earlier management objective may be to simply *arrest* the declining trends in such size metrics. Since the effect size is reduced with the same level of variance it is likely that such an objective would be achieved with fewer years of monitoring. Clearly there is scope to explore power for a range of desired community size states or trends in size metrics.

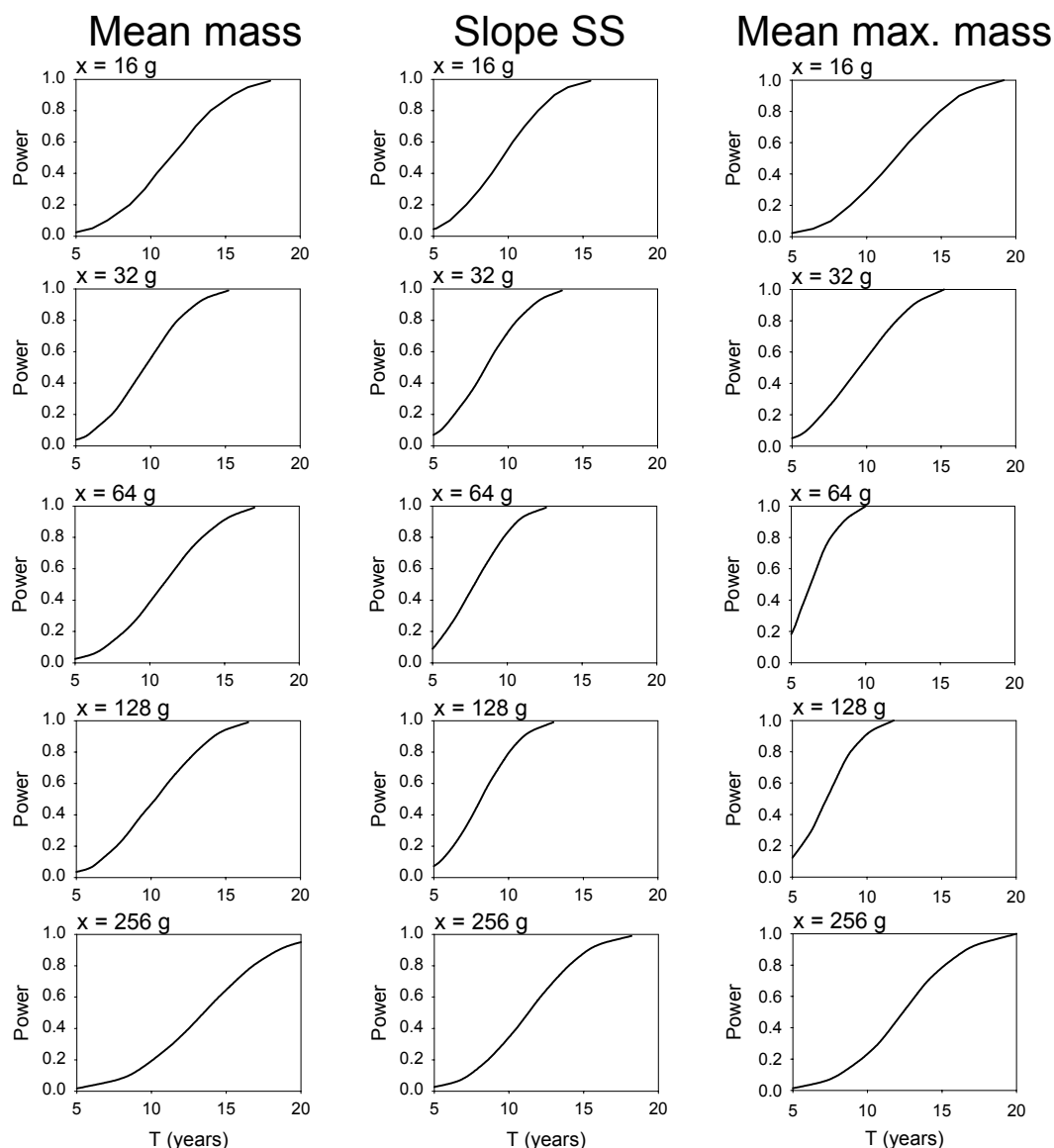


Figure 2.4.4.1. Power to detect temporal trends in the mean mass, slope of size spectrum and mean maximum body mass of North Sea fishes as a function of T . Power for trends in mean mass and the slope of size spectrum was determined for body mass classes x – 4096 g, power for mean maximum mass using all mature fishes of body mass $> x$ g. Expected slopes of trends were based on slopes observed 1982–2000.

2.5 Conclusions

The original EcoQOs that were selected for WGFE to explore were associated with fish community size structure (ICES, 2003). A range of EcoQOs indicators were considered at the meeting along with size-based approaches.

The theory under-pinning size-based approaches is well-developed but further theoretical development that would be useful includes (i) determining the relative impacts of fishing in light of other background variation in the environment, (ii) a reconciliation of macroecological and size spectra theory in order to strengthen the theoretical framework, and (iii) gear catchability by size classes of fish sampled would allow for a better comparison of routine survey data to theoretical reference levels.

Choosing a single indicator or narrow range of indicators (e.g., size-based methods only) may not be desirable, as other metrics may shed light on other aspects of community structure and, therefore, provide complementary information. One approach is to select metrics that are relevant for particular policy drivers or policy questions. A list of potentially useful EcoQOs (as illustrated in Table 2.5.1) may provide a useful framework with which to classify regional and local

EcoQOs, and the associated data required and problems associated with each indicator. It should be noted that such potential indicators would also require evaluation and should have a theoretical underpinning.

An important issue that needs to be addressed regards the setting of reference levels and reference directions, as even in ecosystem-based fisheries management, it may not be realistic to expect a recovery to former ecosystem states, due to the potential for multiple stable states to exist in marine ecosystems. Additionally, the data required to quantify reference levels for the "unexploited" state will be lacking for many regions. More work on the evaluation of ecosystem indicators, their performance and development of reference directions is required (e.g., Rice, 2000, 2003; Rochet and Trenkel, 2003; Trenkel and Rochet, 2003). A potential framework for evaluation of community and ecosystem indicators is signal detection theory and retrospective simulation, as discussed by Rice (2003). Further considerations of the management objectives for EcoQOs are, therefore, required.

Table 2.5.1. Example of table of indicators matched with problem and sampling requirements based on “Potential indicators based on data in COBRA database” (Source: Anon, 2004).

Indicator	Concern – environmental problem - explanation	Sampling
<i>Fish Community</i> (catchable with gillnets)		
No of species by appropriate category - No of threatened species (C?) - No of alien species (C?)	Biodiversity Fisheries	Standardized methods (at community level) to be used in all study areas
Species diversity (C)	Changes in fish fauna	Standardized methods (at community level) to be used in all study areas
Ratio between functional groups - Cyprinid/perchids (C) - Proportion of <i>piscivorous</i> fish (C?) - Benthic/pelagic species (-)	Fisheries Eutrophication Community function Community function	Locally adapted methods Locally adapted methods Locally adapted methods
Size structure of community (C)	Fisheries Environmental conditions	Standardized methods (e.g., same selectivity)
No of specimens – total abundance (C)	Productivity	Locally adapted methods
Total biomass (C)	Productivity	Locally adapted methods
<i>Fish populations</i>		
Abundance - Threatened species/populations (C?) - Sentinel species (C) - Larval abundance (N)	Fisheries Habitat quality Predation and competition Recovery actions	Standardized methods (minimum core set) Locally adapted methods Locally adapted methods
Recruitment - No of juvenile fish (C?) - Ichthyoplankton studies (N)	Habitats Spawning stock Health Climate	Locally adapted methods
Year class strength (C)	Habitats Spawning stock Health Climate	Locally adapted methods
Size and age structure of population (C)	Fisheries Recruitment success/failure	Standardized methods
Sex ratio (C)	Fisheries Hormone disruptions	Locally adapted methods
Age and size at first maturity (C)	Fisheries Climate Environmental conditions	Locally adapted methods
<i>Individual level</i>		
Growth rate (C)	Feeding conditions (longterm) Competition Climate/temperature Salinity	Standardized methods
Liver somatic index (N)	Exposure to contaminants	Locally adapted methods
Gonad somatic index (N)	Exposure to contaminants Climate	Locally adapted methods
Condition factor (C)	Feeding conditions (short-term)	Locally adapted methods
Fat content (N)	Environmental conditions Feeding conditions	Locally adapted methods
Fecundity (N)	Exposure to contaminants Climate	Locally adapted methods
Prevalence of parasites and diseases (C)	Health/stress	Locally adapted methods

C= can be derived from data in COBRA database

N = can be derived from data in national databases

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3 THREATENED AND DECLINING FISH SPECIES

3.1 Review of methods of identifying threatened and declining fish species and critique of the Texel-Faial criteria

3.1.1 Introduction

Fisheries exploitation has resulted in declines in the abundance of target and non-target species to the point where a number of target stocks have been exploited beyond safe biological limits and some non-target species (e.g., skates, angel sharks and sawfishes) have declined to the point of local or regional extinction (Christensen *et al.*, 2003; Dulvy *et al.*, 2003; Myers *et al.*, 1996, 1997; Roberts and Hawkins, 1999). This has led to increased concern over the status of fish populations, including both commercially-important and non-target species, over the last two decades (Myers *et al.*, 1996, 1997; Roberts and Hawkins, 1999; Vincent and Hall, 1996; Wheeler and Sutcliffe, 1990). Most work on the conservation status of fishes has concentrated on freshwater and diadromous species, mainly because such habitats are more amenable to comprehensive study, the threats are more focussed and visible, and several such species appear to be highly threatened (Harrison and Stiassny, 1999; Huntsman, 1994; McDowall, 1992; Reynolds *et al.*, 2002).

Although absolute or global scale fish extinctions have been documented these are primarily for freshwater species, and are a result of several important factors, including habitat destruction and modification, hybridisation, the effects of introduced species, overexploitation, pollution and disease (Angermeier, 1995; Harrison and Stiassny, 1999; Miller *et al.*, 1989; Parent and Schriml, 1995). Marine fishes were traditionally thought to be less vulnerable to extinction, as many species have a wide geographic range, high fecundity and broadcast spawning life history (for both sides of this argument see Dulvy *et al.*, 2003; Huntsman, 1994; Hutchings, 2001a; Musick, 1999; Powles *et al.*, 2000; Roberts and Hawkins, 1999; Sadovy, 2001). In more recent years, however, there has been an increased concern over the status of certain marine fishes, particularly large, slow growing species, species with small geographic ranges and species with high habitat specificity (Barrera Guevara, 1990; Musick, 1999; Reynolds *et al.*, 2001; Sadovy and Cheung, 2003). Additionally, there has been wider concern among conservation biologists and the general public at the impact of fisheries exploitation upon marine biodiversity and the wider ecosystem (Jennings and Kaiser, 1998; Reynolds *et al.*, 2002; Roberts and Hawkins, 1999).

International signatories of the Convention of Biological Diversity are committed to achieving a significant reduction in the rate of biodiversity loss by 2010 in response to the World Summit on Sustainable Development (WSSD, Johannesburg, 2002). This requires the identification and protection of potentially endangered species (Reynolds and Mace, 1999). Typically this has been done using various threat listing protocols, which are largely adapted from terrestrial conservation initiatives. Only recently have these threat listing protocols been applied to marine species and this has resulted in considerable discussion and some disagreement over their validity and utility (Dulvy *et al.*, 2003; Mace and Hudson, 1999; Matsuda *et al.*, 2000; Powles *et al.*, 2000; Reynolds *et al.*, 2002). There is ongoing debate as to whether it is valid to apply threat criteria (e.g., those developed by CITES and IUCN) which have their origins in terrestrial conservation, particularly that of mammals and birds, to some marine organisms. In particular three questions have been raised;

- is it valid to apply decline criteria to commercially exploited species where the range of decline is within management targets? (see Butterworth, 2000; Mace and Hudson, 1999; Punt, 2000).
- are the decline thresholds, which are based on terrestrial taxa, appropriate for threat listing in marine fishes? (see Matsuda *et al.*, 1998; Powles *et al.*, 2000; Punt, 2000).

- do threat categorisations accurately reflect the perceived lower extinction risk of widely distributed, fecund broadcast spawning species which are highly variable in abundance? (see FAO, 2000; Hutchings, 2001a, b; Matsuda *et al.*, 1997; Musick, 1999; Reynolds *et al.*, 2002; Vincent and Hall, 1996).

As an aid to this complex debate, an overview of threat listing protocols and other quantitative methods of assessing extinction risk, focussing primarily on the fisheries literature, is provided below. For these threat listing protocols, the aims, geographical scope, legislative basis, threat listing process, criteria and enforcement are summarised. These criteria are broadly categorised into international (Section 3.1.2), regional (Section 3.1.3) and national threat listing criteria (Section 3.1.4), with emphasis given to those countries bordering the North Atlantic Ocean.

3.1.2 International Fora

3.1.2.1 Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) (<http://www.cites.org/index.html>)

Aims and geographic scope

CITES aims to ensure the international trade in plants and animals does not threaten their survival. Near global; 180 states are currently signatories of CITES.

Legislative basis and links

The Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) was first drafted as a result of a resolution adopted in 1963 at a meeting of the World Conservation Union (IUCN). The Convention was agreed at a meeting of representatives of 80 countries in 1975. The Convention is voluntary agreement by which states, or 'Parties' as they are known, agree to be bound. While states are legally bound to implement CITES, it does not take the place of national laws. Instead CITES provides a framework to be respected by the Parties, which have to adopt their own domestic legislation to ensure national implementation of CITES.

The Parties meet at a Convention of Parties (CoPs) every two to three years to review the implementation of the Convention, and to add species or move them between Appendices. The Conference of Parties has established four permanent committees to facilitate its work; the Standing Committee, Animals Committee, Plants Committee and Nomenclature Committee. The Standing Committee has two main functions, it provides policy guidance on the implementation of the Convention and oversees the Secretariat's budget. In addition it coordinates the work of the other three Committees. The other Committees provide technical advice on appropriate categorisation, unsustainable trade and taxonomy. The secretariat of CITES is administered by the United Nations Environment Programme (UNEP) and plays a coordinating, advisory and servicing role in the working of the Convention.

The categorisation process

CITES works by controlling international trade of species according to the required degree of protection as specified in four Appendices. The assessment process is hierarchical, such that a species is sequentially assessed for listing in Appendix I, then II, III and finally 'no listing' (Table 3.1). Species are added or moved to Appendix I and II only at the Conference of Parties. But for Appendix III species can be added or moved at any time by any Party (Table 3.2).

- **Appendix I** includes species threatened with extinction that are or may be affected by trade. Trade in these species is permitted only in exceptional circumstances.
- **Appendix II** includes species that are not currently threatened with extinction, but that may become so unless trade is closely controlled. This also includes 'look-alike' species that look like other listed species.
- **Appendix III** contains species protected in at least one country, which has asked other CITES Parties for help in controlling the trade.
- **Appendix IV** makes provision for the issue of permits, which are required before international trade in species listed on Appendices I, II and III can occur.

Listing criteria

The Convention itself provides little guidance on how to decide which species to list in the Appendices. This is done in accordance with the criteria laid out in Resolution of the Conventions 9.24 (1994). The criteria rely on trends in species populations and their distributions and were developed along similar lines as the IUCN Red List criteria (Rosser *et al.*, 2001). The criteria have quantitative guidelines, but this does not mean rigorous survey data have to be available for the species to qualify. It is recognised that there is a continuum of extinction risk and trade pressure, so the guideline values are in a sense arbitrary. A species can be listed on Appendix I if it first meets the *Trade* criterion and also meets any one of four *Biological* criteria. The four biological criteria include guidelines and thresholds for small population size, restricted area of distribution, declining populations and a within 5 year criteria (Table 3.2). While the first two criteria explicitly refer to small populations and restricted distributions, this third criterion can theoretically be met by species that are very abundant and widespread, e.g., exploited fishes.

For Appendix II The definition of ‘threatened with extinction’ is the same as for Appendix I but the key distinction is that species are eligible if ‘international trade in all specimens were not subject to strict control’ (Annex 2a) or if listing under Appendix II allows improvement in the control of other threatened Ai or Aii species (i.e., look-alike criteria, Annex 2b).

Appendix III emphasises the legal status of species, an assessment is made as to whether the species is subject to regulation within the jurisdiction of a CITES Party and does effective regulation require the cooperation of other parties.

Those fish species currently listed under CITES are given in Table 3.2

Enforcement

Granting export permits or re-export certificates controls international trade. Permits and certificates are only granted if the relevant authorities are satisfied that trade will not be detrimental to the survival of species in the wild.

Table 3.1. Listing categories (Appendices) of CITES and their definitions.

List category	Definition
Appendix I	Includes all species threatened with extinction, which are or may be affected by trade. Trade in specimens of these species must be subject to particularly strict regulation in order not to endanger further their survival and must only be authorized in exceptional circumstances.
Appendix II	<ul style="list-style-type: none">a) All species which although not necessarily now threatened with extinction may become so unless trade in specimens of such species is subject to strict regulation in order to avoid utilization incompatible with their survival; and;b) other species which must be subject to regulation in order that trade in specimens of certain species referred to in sub-paragraph (a) of this paragraph may be brought under effective control. (i.e., the look alike criteria).
Appendix III	Includes all species which any Party identifies as being subject to regulation within its jurisdiction for the purpose of preventing or restricting exploitation, and as needing the co-operation of other Parties in the control of trade.

Table 3.2. Criteria for inclusion in Appendices of CITES.

Criteria	Definition
Appendix I	A species can be listed if it first meet the <i>Trade</i> criterion and any one of the four <i>Biological</i> criteria
<i>Trade criterion</i>	<p>A species “is or may be affected by trade criterion” and therefore meets the trade criterion if:</p> <ul style="list-style-type: none"> i) it is known to be in trade; or ii) it is probably in trade but conclusive evidence is lacking; or iii) there is potential international demand for specimens; or iv) it would probably enter trade were it not subject to Appendix I controls.
<i>Biological criteria</i>	<p>A species is considered threatened with extinction if it meets or is likely to meet at least one of the following:</p> <p>A wild population is small, and is characterised by at least one of the following:</p> <ul style="list-style-type: none"> i) a decline in number of individuals or the area and quality of habitat, a guideline for a relevant decline is $\geq 20\%$ of the total population within the longer of either 10 years or three generations. A generation is defined as the average age of parents in the population. Importantly a population decrease as the result of a harvesting programme that reduces the population to a predetermined level is not considered a decline. ii) Each sub-population being very small. The guideline for a very small sub-population is < 500 individuals. iii) A majority ($> 50\%$) of individuals, during one or more life history phases, being concentrated in one sub-population. iv) Large ($> \text{one order of magnitude}$) short-term (two years) fluctuations in the number of individuals. v) A high vulnerability due to a species’ biology or behaviours (including migration). Behaviour, such as predictable congregations at particular sites which make a species particularly easy to harvest may also be considered vulnerable.
B: Restricted area of distribution	<p>The wild population has a restricted area of distribution ($< 10,000 \text{ km}^2$) and is characterised by at least one of the following:</p> <ul style="list-style-type: none"> i) fragmentation or occurrence at very few locations, similar to Aii. ii) Large fluctuations in the area of distribution or the number of sub-populations, similar to Aiv but without reference to a time scale. iii) A high vulnerability due to a species’ biology or behaviour (including migration), similar Av. iv) A decrease in any one of: the area of distribution, the number of sub-populations, the number of individuals, the area or quality of habitat or reproductive potential, similar Ai.
C: Decline of wild population	<p>A decline ($\geq 50\%$ in the greater of either 5 years or two generations) in the number of individuals in the wild, which has been either:</p> <ul style="list-style-type: none"> i) observed as ongoing or as having occurred in the past (but with the potential to resume). ii) Inferred or projected on the basis of any one of the following: a decrease in area or quality of habitat, levels or patterns of exploitation, threats from extrinsic factors such as the effects of pathogens, competitors, parasites, predators, hybridisation, introduced species and the effects of toxins and pollutants, decreasing reproductive potential. <p>While the first two criteria explicitly refer to small populations and restricted distributions, this third criterion can theoretically be met by species that are very abundant and widespread.</p>

Criteria	Definition
D: Within 5 years	The fourth criterion is met if the species does not currently meet any of the three biological criteria, but there is sufficient reason to believe it is likely to do so within the next five years.
Appendix II	The biological criteria are the same as for Appendix I. This Appendix is split into two parts, a and b
2a	<p>a) A species is listed if it is known, inferred or projected that unless trade in the species is subject to strict regulation, it will meet at least one of the biological criteria for inclusion in Appendix I in the near future or</p> <p>b) It is known, inferred or projected that harvesting of specimens from the wild for international trade has, or may have, a detrimental impact on the species by either (i) exceeding, over an extended period, the level that can be continued in perpetuity; or (ii) reducing it to a population level at which its survival would be threatened by other influences</p>
2b	<p>‘look-alike criteria’</p> <p>a) the specimens resemble those of a species included in Appendix II [1] or in Appendix I such that a non-expert, with reasonable effort, is unlikely to be able to distinguish between them; or</p> <p>b) the species is a member of a taxon of which most of the species are already included in Appendix II [1] or in Appendix I, and the remaining species must be included to bring trade in specimens of the others under effective control.</p>
Appendix III	<p>When considering species for inclusion in Appendix III a Party:</p> <p>a) ensure that:</p> <ul style="list-style-type: none"> i) the species is native to its country; ii) its national regulations are adequate to prevent or restrict exploitation and to control trade, for the conservation of the species, and include penalties for illegal taking, trade or possession and provisions for confiscation; and iii) its national enforcement measures are adequate to implement these regulations; <p>b) determine that, notwithstanding these regulations and measures, there is indication that the co-operation of the Parties is needed to control trade.</p> <p>c) inform the Management Authorities of other range States, the known major importing countries, the Secretariat and the Animals Committee of the Plants Committee that it is considering the inclusion of the species in Appendix III and seek their opinion on the potential effects of such inclusion; and</p> <p>d) after due consultation, and having satisfied itself that the biological status and trade status of the species justify the action, submit to the Secretariat the name of the species it wishes to include in Appendix III</p>

Footnote

1. under the provisions of Article II, paragraph 2 (a) of the Convention.

Table 3.3. List of fishes protected under CITES.

Family	Species	Appendix
Rhincodontidae	¹ Whale shark <i>Rhincodon typus</i>	II
Cetorhinidae	¹ Basking shark <i>Cetorhinus maximus</i>	II
Lamnidae	^{1,2} White shark <i>Carcharodon carcharias</i>	III
Acipenseridae	¹ Shortnose sturgeon <i>Acipenser brevirostrum</i>	I
	¹ Atlantic sturgeon <i>Acipenser sturio</i>	I
	³ Acipenseriformes (other species)	II
Osteoglossidae	Arapaima <i>Arapaima gigas</i>	II
	Asian arowana <i>Scleropages formosus</i>	I
Cyprinidae	African blind barb fish <i>Caecobarbus geertsi</i>	II
	Ikan temoleh <i>Probarbus jullieni</i>	I
Catostomidae	Cui-ui <i>Chasmistes cujus</i>	I
Pangasiidae	⁴ Giant catfish <i>Pangasianodon gigas</i>	I
Syngnathidae	¹ Seahorses <i>Hippocampus</i> spp. (from May 2004)	II
Sciaenidae	¹ Totoaba <i>Totoaba macdonaldi</i>	I
Latimeriidae	¹ Coelacanth <i>Latimeria</i> spp.	I
Ceratodontidae	Australian lungfish <i>Neoceratodus forsteri</i>	II

Footnotes

¹. Occur in marine waters;². Australia;³. Sturgeons and paddlefish, some of which occur in marine and estuarine waters;⁴. may occasionally occur in brackish water**3.1.2.2 World Conservation Union (IUCN) Red list of threatened species (www.iucn.org)***Aims and geographical scope*

The mission of the World Conservation Union (IUCN) is to influence, encourage and assist societies throughout the world to conserve the integrity and diversity of nature and to ensure that any use of natural resources is equitable and ecologically sustainable. The goals of the IUCN Red List Programme are to identify and document those species most in need of conservation attention if global extinction rates are to be reduced, and provide a global index of the state of degeneration of biodiversity.

Legislative basis and links

The World Conservation Union does not have a legislative mandate, but instead has more of a coordinating role. The IUCN brings together 76 states, 111 government agencies, 720 NGOs, 35 affiliates, and some 10,000 scientists and experts from 181 countries worldwide. It does this through 6 commissions, one of these the Species Survival Commission produces a Red List of Threatened Species. This approach has become a world standard (Groombridge and Jenkins, 2000).

The categorisation process

The species survival commission consists of a large number of Species Specialist Groups, which are largely taxon-specific. In most cases the Red List Authorities is the Species Specialist Group responsible for individual taxonomic groups, and their role is to ensure that all species within their jurisdiction are correctly evaluated at least once every ten years and if possible every 5 years. Volunteer scientists and specialists coordinated by these bodies assess taxa. These assessments are evaluated by at least two members of the Red List Authority. Current Specialist Groups (SGs) dealing with marine fishes include the Grouper and Wrasse SG, Salmon SG, Shark SG and Sturgeon SG.

The listing scheme

The IUCN Red List is one of the most detailed threat listing schemes. It lists species in one of 9 semi-hierarchical categories (<http://www.redlist.org>). Species are considered for listing as *Extinct*, *Extinct in the wild*, or as one of the *threatened* categories (including *Critically Endangered*, *Endangered*, *Vulnerable*), *Near threatened* and *Least Concern*. Below this level species are recorded as *Data deficient* or *Not evaluated* (Table 3.4).

A species is *Extinct* when there is no reasonable doubt that the last individual has died. A taxon is *Extinct in the Wild* when it survives only in cultivation, in captivity or as a naturalised population well outside its previous range. Five criteria are used to assess whether a species is *Threatened*, if a species does not qualify but is close to qualifying or likely to qualify in the near future it is categorised as *Near Threatened*. If there is adequate information and a taxon fits none of these categories it is designated *Least Concern*.

Listing criteria

There are five criteria (A-E) with numerous sub-criteria and definitions used to assess whether a species meets one of the three *Threatened* criteria. These criteria can broadly be summarised as:

- a) reduction in population size;
- b) reduction in geographic range;
- c) small population size (<10 000 mature individuals) which is also declining;
- d) very small population size (estimated to be fewer than 1000 mature individuals) or very restricted distribution (< five regional areas); and
- e) quantitative analysis that indicates a high probability of extinction over a defined time scale.

There are a number of issues and problems that are explicitly recognised in the IUCN Red List documentation including data quality, spatial scale and uncertainty (IUCN, 2001). The criteria are quantitative and it is recommended that the absence of high quality data should not deter attempts to apply the criteria. The use of estimation, inference and projection are acceptable, so long as they can be supported and specified as part of the documentation. The size of geographic ranges and patterns of occupancy are dependent upon the spatial scale at which they are measured. At larger scales ranges estimates that are more likely to exceed the thresholds for the threatened categories because mapping will reveal fewer unoccupied areas compared with mapping at finer scales. While this issue is a recognised source of inconsistency and the Red List documentation acknowledges that the most appropriate scale will depend on the taxon and the quality of the distributional data. Uncertainty can arise from natural variation, vagueness of terms and definitions and measurement error. Where uncertainty leads to a variety of possible assessment outcomes then it is recommended that a single category be chosen on the basis that it is precautionary and credible, and that the decision is documented.

There is ongoing debate as to whether it is valid to apply IUCN Red List criteria, which have their origins in terrestrial conservation, particularly that of mammals and birds, to some marine organisms. In particular two questions have been raised;

- is it valid to apply decline criteria to species where the range of decline is within management targets? (see Mace and Hudson, 1999).
- do the threat categorisations accurately reflect the perceived lower extinction risk of highly variable widely distributed, fecund broadcast spawning species? (see Hutchings, 2001a; Matsuda *et al.*, 1997; Musick, 1999; Reynolds *et al.*, 2002; Vincent and Hall, 1996).

This debate is complex and unresolved and in response to this the thresholds for the decline in population size (A1–4) have been revised in more recent versions and a marine caveat was devised and has been attached to listing of a small number of marine fishes (Table 3.4).

Table 3.4. World Conservation Union IUCN criteria 2001 (version 3.1) and COSEWIC criteria for determining the status of threatened species.

Category	IUCN		COSEWIC		
	Critically endangered	Endangered	Vulnerable	Endangered	Threatened
A. Reduction in population size This may be a past population decline [1], or suspected future decline, based on direct observations, appropriate index of abundance, decline in area of occupancy, extent of occurrence and/or quality of habitat, actual or potential levels of exploitation and the effects of introduced taxa, hybridisation, pathogens, pollutants, competitors or parasites.					
1. Population decline rate [1] over the longer of 10 years or 3 generations [2] based on [3]	≥90%	≥70%	≥50%		
2. Population decline rate [1] over the longer of 10 years or 3 generations [4] based on [3]	≥80%	≥50%	≥30%	≥50%	≥20%
3. Projected or suspected population reduction within the next 10 years or three generations based on [3]	≥80%	≥50%	≥30%		
4. Population reduction [1] over any 10 year or three generation period, including both the past and future time period [4] based on [3]	≥80%	≥50%	≥30%		
B. Geographic range					
1. Extent of occurrence, and estimates indicating at least two of a-c	≤100 km ²	<5,000 km ²	<20,000 km ²	<5,000 km ²	<20,000 km ²
2. Area of occupancy, and estimates indicating at least two of a-c	≤10 km ²	<500 km ²	<2,000 km ²	<500 km ²	<2,000 km ²
a. Fragmented or number of isolated sub-populations is	1	≤5	≤10	≤5	≤10
b. Continuing decline [1] in any of: extent of occurrence; area of occupancy; area, extent or quality of habitat; number of locations or sub-populations; number of mature individuals					
c. Extreme fluctuations in any of: extent of occurrence; area of occupancy; number of locations or sub-populations; number of mature individuals					
C. Small population size Based on a combination of population size AND either (1) or (2)					
Population size estimated to be	<250	<2,500	<10,000	<2,500	<10,000
1. Estimated continuing decline	25% in the longer of 3 years or 1 generation	20% in the longer of 5 years or 2 generations	10% in the longer of 10 years or 3 generations	20% in the longer of 5 years or 2 generations	10% in the longer of 10 years or 3 generations
2. Continuing decline [1] in numbers of mature individuals AND at least one of the following					
a (i) no subpopulation containing	>50 mature individuals	>250 mature individuals	>1,000 mature individuals	>250 mature individuals	>1,000 mature individuals
a (ii) OR mature individuals in subpopulation	90%	95%	100%		
b. Extreme fluctuations in number of mature individuals					
D. Very small population size In the form of either					

Category	IUCN		COSEWIC		
	Critically endangered	Endangered	Vulnerable	Endangered	Threatened
1. Number of mature individuals	<50	<250	<1,000	<250	<1,000
2. Population restricted	—	—	Area of occupancy <20 km ² or <5 locations		
E. Quantitative analysis					
Probability of extinction in the wild is at least	50% in 10 years or 3 generations	20% in 20 years or 5 generations	10% in 100 years	20% in 20 years or 5 generations	10% in 100 years
F. Rescue effect adjustment (COSEWIC only): if there is significant and on-going immigration from cross-border populations, downlist one category					
IUCN Marine caveat: The criteria (A-D) provide relative assessments of trends in the population status of species across many life forms. However, it is recognised that these criteria do not always lead to equally robust assessments of extinction risk, which depend upon the life history of a species. The quantitative criterion (A1abd) for the threatened categories may not be appropriate for assessing the risk of extinction for some species, particularly those with high reproductive potential, fast growth and broad geographic ranges. Many of these species have high potential for population maintenance under high levels of mortality, and such species might form the basis for fisheries					

Footnotes

1. observed, estimated, inferred or suspected
2. where the causes of the reduction are clearly reversible AND understood AND ceased.
3. (a) direct observation; (b) an index of abundance appropriate to the taxon; (c) a decline in the area of occupancy, extent of occurrence and / or quality of habitat; (d) actual or potential levels of exploitation; (e) the effect of introduced taxa, hybridisation, pathogens, pollutants, competitors or parasites.
4. where the causes of the reduction or its causes may not have ceased OR may not be understood OR may not be reversible.

3.1.3 Regional Fora

3.1.3.1 Convention for the Protection of the Marine Environment of the North-East Atlantic "OSPAR Convention" (<http://www.ospar.org>)

Aims and geographic scope

The OSPAR Convention includes annexes for the assessment of the quality of the marine environment (Annex IV) and for the protection and conservation of the ecosystems and biological diversity of the maritime area (Annex V). Annex V requires that contracting parties “take the necessary measures to protect and conserve the ecosystems and the biological diversity of the maritime area, and to restore, where practicable, marine areas which have been adversely affected”. There do not appear to be explicit aims for the actual listing criteria. However, the outcomes of the listing process are nested within a framework that ensures appropriate action is taken in the form of either:

- OSPAR (Priority) Action, where OSPAR has the competence to adopt programmes or measures necessary for protection, conservation or where practical restoration. If OSPAR is not competent it should inform the competent authority or international body
- OSPAR Inform, where OSPAR informs the relevant international fora of concern
- Research, where OSPAR should indicate the need for research to determine what actions are required for protection, conservation or where practical restoration to the appropriate authority or body.

The area covered by OSPAR extends from the east coast of Greenland to the continental North Sea coast, southwards to the Straits of Gibraltar and northwards to North Pole. This area covers the Arctic and Atlantic Oceans north of 36°N, and east of 44°W, excepting that area south of 59°N and between 42 and 44°N, and the Mediterranean and Baltic Seas. The OSPAR area is divided into five regions: Region I: The Arctic; Region II: The Greater North Sea; Region III: The Celtic Seas; Region IV: The Bay of Biscay and Region V: The wider Atlantic.

Legislative basis and links

The OSPAR Convention for the Protection of the Marine Environment of the North East Atlantic came into force on 25 March 1998, partly replacing the Oslo Convention (1972) and Paris Convention (1974). The 16 contracting parties are Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, the United Kingdom and the Commission of the European Communities. A Commission consisting of representatives of the contracted parties supervises the implementation of the Convention.

The process

Nominations are presented to the Biodiversity Committee (BDC) of the OSPAR Commission. After agreement by the BDC, the OSPAR commission arranges peer review of the evidence, e.g., by working groups of the International Council for the Exploration of the Sea (ICES, 2002, 2003a). On the basis of this review a preparatory working group of the BDC make a recommendation to the BDC, the BDC then examine this recommendation and if accepted it is presented to the OSPAR Commission for final adoption (Anonymous, 2003).

Listing criteria

For assessing threatened species and habitats, OSPAR use the Texel-Faial criteria. Some of the criteria do not relate to threat *per se* but are used to highlight taxa of significant importance within the OSPAR region. The selection criteria for species are: *Global importance*, *Regional importance*, *Rarity*, *Sensitivity*, *Keystone species* and *Decline* (Table 3.5). Listing is undertaken at the level of species and does not presently offer scope for the separate consideration of stocks or populations.

Table 3.5. The Texel-Faial selection criteria for the identification of species requiring protection.

	Selection criteria for species	Guidance
Global importance	Importance for the OSPAR Area for a species. Importance on a global scale, of the OSPAR Area, for a species when a high proportion of a species at any time of the life cycle occurs in the OSPAR Area.	'High proportion' is considered to be more than 75% when known.
Regional importance	Importance within the OSPAR Area, of the regions for the species where a high proportion of the total population of a species within the OSPAR Area for any part of its life cycle is restricted to a small number of locations in the OSPAR Area.	'High proportion' is considered to be 90% of the population in a small number of locations of 50 km x 50 km grid squares. [1,2]
Rarity	A species is rare if the total population size is small. In case of a species that is sessile or of restricted mobility at any time of its life cycle, a species is rare if it occurs in a limited number of locations in the OSPAR Area, and in relatively low numbers. In case of a highly mobile species, the total population size will determine rarity.	'A limited number of locations' could be in a small number of 50 km x 50 km grid squares, [1,2]. Species which are present in high abundance outside of the OSPAR Area and only occur at the edges of the OSPAR Area will not generally qualify as 'rare' species
Sensitivity	A species is very sensitive when: a. it has very low resistance [3]; and / or b. it has a very low resilience [3], recovery is likely to be achieved only over a very long period, or is likely not to be achieved at all) A species is 'sensitive' when: a. it has low resistance [3]; and / or b. it has low resilience [3], recovery is likely to be achieved only over a very long period, or is likely not to be achieved at all).	'A very long period' may be considered to be more than 25 years and 'long period' in the range of 5 to 25 years. The time frame should be on an appropriate scale for that species. Sensitivity to human activities is measured by: a. life history characteristics (including natural behaviour); b. dependence on other specific ecological attributes e.g., restricted/specific habitat requirements
Keystone species	A species which has a controlling influence on a community	No guidance

	Selection criteria for species	Guidance
Decline	Means an observed or indicated significant decline in numbers, extent or quality (quality refers to life history parameters). The decline may be historic, recent or current. 'Significant' need not be in a statistical sense.	<p>'Decline is divide into the following categories:</p> <ol style="list-style-type: none"> 1. Extirpated (extinct from the OSPAR Area): a population of a species formerly occurring in the maritime area is defined as extirpated: <ul style="list-style-type: none"> • if it was still occurring in the area at any time in the last 100 years; and • if there is a high probability, or it has been proved, that the last individuals have since died or moved away; or • if surveys in the area have repeatedly failed to record a living individual in its former range and/or known or expected habitats at appropriate times (taking into account diurnal, seasonal, annual patterns of behaviour for at least 10 years) 2. Severely declined: a population of species occurring in the maritime area is defined as severely declined <ul style="list-style-type: none"> • if individual numbers show an extremely high and rapid decline in the area over an appropriate time frame, or the species has already disappeared from the major part of its former range in the area; or • if individual numbers are at a severely low level due to a long continuous and distinct general decline in the past. 3. Significantly declined: means a considerable decline in number, extent or quality 4. High probability of a significant decline in number, extent or quality in the future. <p>Where the decline is "clear and present", and can be linked directly or indirectly to human activity, the species is also considered to be "currently threatened". Where there is a high probability of significant decline linked directly or indirectly to human activity, the species is considered to be "potentially threatened". Where the species satisfies criterion 3 (rarity) or 4 (sensitivity), a lower threshold of probability can justify regarding the species as "potentially threatened".</p> <p>For these purposes, "decline" should only be regarded as occurring where the decline goes beyond that which can be expected from what is known about long-term natural variability and resilience in that species, over a time frame appropriate for it.</p>

Footnotes

1. This is dependent on scientific judgment regarding natural abundance, range or extent and adequacy of recording.
2. A different scale may be needed for different taxa.
3. that is, it is very easily adversely affected by human activity.

3.1.3.2 Convention on the Protection of the Marine Environment of the Baltic Sea Area – “the Helsinki Convention” (<http://www.helcom.fi>)

Aims and geographic scope

The Helsinki Commission (HELCOM) is the governing body of the Helsinki Convention and works to protect the marine environment of the Baltic Sea from all sources of pollution and to restore and safeguard its ecological balance. The Convention covers the whole of the Baltic Sea area, which for the purposes of this Convention is the Baltic Sea and the entrance to the Baltic Sea bounded by the parallel of the Skaw in the Skagerrak at 57°44.43'N.

The most important threats are thought to be eutrophication and hazardous substances (e.g., pesticides, heavy metals, PCBs and dioxins). Other identified significant threats include habitat destruction, the use of certain fishing techniques and the presence of non-native species.

Legislative basis and links

The first version of the Helsinki Convention came into force in 1974 signed by the then 7 Baltic Coastal states. In the light of political changes and developments in international environmental and maritime law a new convention was signed in 1992. After ratification the convention came into force on 17 January 2000. The contracting parties are Denmark, Estonia, the European Community, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden.

The process

HELCOM unanimously adopts recommendations of the marine environment that the governments of the Contracting Parties must act on in their respective national programmes and legislation.

The HELCOM HABITAT subsidiary is responsible for HELCOM's recommendations on threatened species, habitats and protected areas. So far the only recommendations concerning threatened species relate to seals (recommendation 9/1 1988), harbour porpoise (recommendation 17/2 1996) and wild salmon *Salmo salar* (recommendation 19/2 1998) and there are no criteria for which species are to be considered threatened. Instead the HELCOM HABITAT promotes ecosystem-based approaches and focus on protecting areas rather than species.

In 1995 62 marine and coastal areas, considered especially ecologically valuable, were designated to establish a system of Coastal and Marine Baltic Sea Protected Areas (BSPAs). Contracting parties were urged to establish management plans for each BSPA to ensure nature protection and sustainable use of natural resources (HELCOM Recommendation 15/5). Many BSPAs within the EU countries have also been designated for the NATURA 2000 network (see Section 3.1.3.4), which entails management obligations. So far very few of the 62 designated BSPAs are fully implemented and protected.

Sites for protection include: 1) areas with high biodiversity, 2) habitats of endemic, rare or threatened species and communities of fauna and flora, 3) habitats of migratory species, 4) nursery and spawning areas, 5) rare or unique or representative geological or geomorphological structures or processes. Furthermore the areas shall be relatively natural, i.e., non-disturbed by man, relatively pollution free, and representative for a Baltic Sea region or state. Contracting parties can apply to HELCOM for designation of additional areas to BSPA's.

In 1998, HELCOM compiled a red list of marine and coastal biotopes that were considered threatened by human activities in the Baltic Sea, the Belt Sea and Kattegat. In 2000 the contracting parties were asked to make sure that heavily endangered or immediately threatened marine and coastal biotopes in the Baltic Sea area were legally protected (Recommendation 21/4).

Apart from the recommendations HELCOM manages two projects concerning threatened fish species. A captive breeding project has started to conserve Baltic sturgeon, which is at risk from extinction, and, in cooperation with the International Baltic Sea Fishery Commission (IBSFC), HELCOM has made efforts to protect and restore wild salmon populations.

3.1.3.3 Convention on the conservation of European wildlife and natural habitats ‘Bern Convention’ (<http://conventions.coe.int/Treaty/EN/CadreListeTraites.htm>)

Aims and geographical scope

The Convention has three aims; to conserve wild flora and fauna and natural habitats; to promote co-operation between States; and to give particular attention to endangered and vulnerable species, including endangered and vulnerable migratory species (Article 1).

Legislative basis and links

In 1979, the Council of Europe (comprised of 45 member states) adopted the Convention on the conservation of European wildlife and natural habitats, known as the ‘Bern Convention’. The member states or Parties are represented at a Standing Committee established by the Convention. The Committee's principal task is to monitor the provisions of this Convention and it is especially competent to make recommendations to the Parties and amendments to the appendices (Table 3.6).

The process

The Convention does not directly undertake conservation assessment but instead sets out obligations of Contracting Parties (nations). The Parties are obliged to; promote national conservation policies; have regard for conservation in regional planning policies and pollution and promote education and information. Signatories can undertake this by implementing legislation at a national level (e.g., see Wildlife and Countryside Act 1981) and by proposing amendments to the Appendices. However toward the aims of the Convention, Parties are obliged to ensure the special protection of the fauna and flora listed in the appropriate Appendices (Table 3.6). Groups of Experts submit guidelines, recommendations and action plans to the Standing Committee, which meets annually.

Threat Criteria

The Convention uses an older version of World Conservation Union (IUCN) Red List criteria in order to bring the terminology into line with CITES. The categories include *Endangered*, *Vulnerable*, *Rare* and *Threatened* (Table 3.7). The Convention is not explicit on the criteria or guidelines for the inclusion of species on its appendices, particularly Appendix III. The species of marine, estuarine and diadromous fishes currently listed in Appendices II and III of the Bern convention are summarised in Table 3.8.

Table 3.6. Listing categories (Appendices) used by the Bern Convention and their description

List category	Definition
Appendix I	Strictly protected flora species, based on the list of species endangered in the region which are to be specially protected
Appendix II	Strictly protected fauna species, based on the list of species (both vertebrates and invertebrates) endangered in the region which are to be specially protected
Appendix III	Protected fauna species, appearing in Appendix III are to be protected, but exploitation is possible if the population level permits.
Appendix IV	This lists prohibited means and methods of killing, capture and other forms of exploitation

Table 3.7. Definitions used by the Bern Convention.

Threat criteria	Definition
Endangered	Taxa in danger of extinction and whose survival is unlikely if the causal factors continue operating. Included are taxa whose numbers have been reduced to a critical level or whose habitats have been so drastically reduced that they are deemed to be in immediate danger of extinction.
Vulnerable	Taxa believed likely to move into the endangered category in the near future if the causal factors continue operating. Included are taxa of which most or all the populations are <i>decreasing</i> because of over-exploitation, extensive destruction of habitat or other environmental disturbance; taxa with populations that have been seriously <i>depleted</i> and whose ultimate security is not yet assured; and taxa with populations that are still abundant but are <i>under threat</i> from serious adverse factors throughout their range.
Rare	Taxa with small world populations that are not at present endangered or vulnerable, but are at risk. These taxa are usually localised within restricted geographical areas or habitats or are thinly scattered over a more extensive range.
Threatened	Threatened is used in the conservation context for species which are in one of the three categories endangered, vulnerable or rare. Species are marked as threatened where it is <i>known</i> that they are endangered, vulnerable or rare, but there is not enough information to say which of the three categories is appropriate.

Table 3.8. Marine and diadromous fishes listed under the Bern Convention. Those freshwater fishes that are occasionally recorded from brackish waters are excluded.

Family	Species	Appendix	
		II	III
Petromyzonidae	Lampern <i>Lampetra fluviatilis</i>	-	yes
	Sea lamprey <i>Petromyzon marinus</i>	-	yes
Squatinae	Angel shark <i>Squatina squatina</i> [1]	-	yes
Cetorhinidae	Basking shark <i>Cethorinus maximus</i> [1]	yes	-
Lamnidae	White shark <i>Carcharodon carcharias</i> [1]	yes	-
	Mako <i>Isurus oxyrinchus</i> [1]	-	yes
	Porbeagle <i>Lamna nasus</i> [1]	-	yes
Carcharhinidae	Blue shark <i>Prionace glauca</i> [1]	-	yes
Rajidae	White skate <i>Rostroraja (Raja) alba</i> [1]	-	yes
Mobulidae	Devil fish <i>Mobula mobular</i> [1]	yes	-
Acipenseridae	Adriatic sturgeon <i>Acipenser naccarii</i>	yes	-
	Starry sturgeon <i>Acipenser stellatus</i>	-	yes
	Sturgeon <i>Acipenser sturio</i> [1]	yes	-
	Beluga <i>Huso huso</i> [1]	yes	-
	Beluga <i>Huso huso</i>	-	yes
Clupeidae	Allis shad <i>Alosa alosa</i>	-	yes
	Twaite shad <i>Alosa fallox</i>	-	yes
	Pontic shad <i>Alosa pontica</i>	-	yes
Salmonidae	Whitefish <i>Coregonus</i> spp.	-	yes
	Salmon <i>Salmo salar</i>	-	yes
Gasterosteidae	Southern ninespine stickleback <i>Pungitius (Tuntitius) platygaster</i>	-	yes
Syngnathidae	Short-snouted seahorse <i>Hippocampus hippocampus</i> [1]	yes	-
	Seahorse <i>Hippocampus ramulosus</i> [1]	yes	-
	Black-striped pipefish <i>Syngnathus abaster</i>	-	yes
Cottidae	Fourhorn sculpin <i>Myoxocephalus quadricornis</i>	-	yes
Serranidae	Dusky grouper <i>Epinephelus marginatus</i> [1]	-	yes
Sciaenidae	Brwon meagre <i>Sciaena umbra</i> [1]	-	yes
	Shi drum <i>Umbrina cirrosa</i> [1]	-	yes
Gobiidae	<i>Knipowitschia (Padogobius) panizzae</i>	-	yes
	Bighead goby <i>Neogobius (Gobius) kessleri</i>	-	yes
	Syrman goby <i>Neogobius (Gobius) syrman</i>	-	yes
	<i>Pomatoschistus canestrinii</i> [1]	yes	-
	<i>Pomatoschistus canestrinii</i> [1]	-	yes
	<i>Pomatoschistus microps</i>	-	yes
	Sand goby <i>Pomatoschistus minutus</i>	-	yes
	<i>Pomatoschistus tortonesei</i> [1]	yes	-
	Tubenose goby <i>Proterorhinus marmoratus</i>	-	yes
	Grass goby <i>Zosterisessor (Gobius) ophiocephalus</i>	-	yes

Footnote

1. only applies in the Mediterranean Sea

3.1.3.4 EU Habitats Directive (<http://europa.eu.int/comm/environment/nature/habdir.htm>)

Aims and geographical scope

The EU Habitats Directive (92/43/EEC) on the conservation of natural habitats and of wild flora and fauna aims at the promotion of “the maintenance of biodiversity, taking account of economic, social, cultural and regional requirements”, as “a contribution to the general objective of sustainable development”; “whereas, in the European territory of the Member States, natural habitats are continuing to deteriorate and an increasing number of wild species are seriously threatened; whereas given that the threatened habitats and species form part of the Community's natural heritage and the threats to them are often of a transboundary nature, it is necessary to take measures at Community level in order to conserve them.”

Geographically, the EU Habitats Directive applies for the member states of the European Union (Austria, Belgium, Denmark, Germany, Finland, France, Greece, Ireland, Italy, Luxemburg, Netherlands, Portugal, Spain, Sweden, United Kingdom). On 1 May 2004, the Habitats Directive will be extended to the new member states (including Cyprus, Estonia, Latvia, Lithuania, Malta and Poland).

The EU Habitats Directive aims in setting up a coherent European ecological network of special areas of conservation under the title Natura 2000. “This network, composed of sites hosting the natural habitat types listed in Annex I and habitats of the species listed in Annex II, shall enable the natural habitat types and the species' habitats concerned to be maintained or, where appropriate, restored at a favourable conservation status in their natural range.” (Art. 3.1).

Legislative basics and links

The EU Habitats Directive was adopted on 21 May 1992. The directive was amended by the accession act of Austria, Finland and Sweden on 1 Jan. 1995. In October 1999, the directive was supplemented with the “Interpretation Manual of European Habitats EUR 15/2” which is intended to give a guideline for the interpretation of names in case of classification difficulties. (see: <http://europa.eu.int/comm/environment/nature/hab-en.htm>)

The Habitats Directive is now linked with the EU Water Framework Directive (2000/60/EC), which was adopted on 23 October 2000.

The categorisation process

The EU Habitats Directive includes 6 annexes listing relevant items for the conservation of habitats and species.

Annex 1 lists “natural habitat types of community interest whose conservation requires the designation of special areas of conservation.” It includes a number of marine habitats, including priority habitats (indicated with an asterix):

- 1110 Sandbanks which are slightly covered by sea water all the time
- 1120 * *Posidonia* beds (*Posidonium oceanicae*)
- 1130 Estuaries
- 1140 Mudflats and sandflats not covered by seawater at low tide
- 1150 *Coastal lagoons
- 1160 Large shallow inlets and bays
- 1170 Reefs
- 1180 Submarine structures made by leaking gases,

Annex 2 comprises “animal and plant species of community interest whose conservation requires the designation of special areas of conservation,” including the relevant anadromously migrating and euryhaline fish species.

Lampetra fluviatilis (except the Finnish and Swedish populations)

Petromyzon marinus (except the Swedish populations)

*Acipenser sturio**

Alosa spp.

*Coregonus oxyrhynchus** (anadromous populations in certain sectors of the North Sea)

Aspius aspius (except the Finnish populations)

Cobitis taenia (except the Finnish populations)

Misgurnus fossilis

Cottus gobio (except the Finnish populations)

Asterisks indicate priority species. The names in the list are not taxonomical but jurisdictional units; in the case of subsequent taxonomic changes, the names in the list are not amended as the list refers to the taxonomical status of 1992. Anadromously migrating species with parts of their life cycles in fully marine water include *Lampetra fluviatilis*, *Petromyzon marinus*, *Acipenser sturio*, *Alosa* spp. and *Coregonus oxyrhynchus*. Euryhaline species living in brackish water of parts of the Baltic Sea include *Aspius aspius*, *Cobitis taenia*, *Misgurnus fossilis*, and *Cottus gobio*.

Annex 4 lists “animal and plant species of community interest in need of strict protection.” It includes two priority fish species, *Acipenser sturio* and *Coregonus oxyrhynchus* (anadromous populations in certain sectors of the North Sea).

Annex 5 addresses “animal and plant species of community interest whose taking in the wild and exploitation may be subject to management measures”, and includes *Alosa* spp.

Listing criteria

There are no formal criteria for including habitats and species in Annexes 1, 2, and 4. The existing lists of habitats species were compiled by expert groups prior to 1992, and then formally politically adopted. At present, habitats or species cannot be added to or removed from the lists. As an exception, when new member states are accessed, the lists of habitats and species are updated. There are future plans to improve the coverage of marine habitats and species in the Habitats Directive.

In Annex 3, criteria for listing habitats and species conservation areas are given. For species, the following criteria are relevant:

- a) Size and density of the population of the species present on the site in relation to the populations present within national territory.
- b) Degree of conservation of the features of the habitat which are important for the species concerned and restoration possibilities.
- c) Degree of isolation of the population present on the site in relation to the natural range of the species.
- d) Global assessment of the value of the site for conservation of the species concerned.

3.1.4 National Fora

3.1.4.1 Wildlife and Countryside Act 1981 Great Britain

Aim and geographic scope

This is the major legal instrument for wildlife protection in Great Britain, therefore excluding Northern Ireland, Channel Islands or the Isle of Man. Wildlife protection in Northern Ireland and the Isle of Man is provided by the following acts: Wildlife (Northern Ireland) Order 1985, and the Wildlife Act 1990 (IoM). The protection afforded only extends throughout Great Britain unless otherwise specified, and to adjacent territorial waters, which currently extend 12 miles out to sea.

Legislative basis and links

This legislation is the means by which the Convention on the Conservation of European Wildlife and Natural Habitats (the 'Bern Convention') and the European Union Directives on the Conservation of Wild Birds (79/409/EEC) and Natural Habitats and Wild Fauna and Flora (92/43/EEC) are implemented in Great Britain.

The listing process

Every five years the Joint Nature Conservation Committee (JNCC) advises Government on which animals and plants should be legally protected by listing on Schedules of the Wildlife and Countryside Act, 1981 in a Quinquennial Review. Species meeting the criteria are listed on 'schedules'. The Act consists of four 'Parts'; Part I is concerned with the protection of wildlife, and the other Parts II-IV relate to the countryside and national parks and the designation of protected areas, public rights of way and miscellaneous provisions of the Act. Part I consists of 27 sections, but the only section of relevance here is Section 9, which prohibits a wide range of activities and practices including the intentional killing, injuring or taking, the possession and the trade in wild animals listed on Schedule 5.

Criteria for species selection

Under Section 22 of the Act, species can be added to the appropriate schedule when one or both of the following circumstances apply:

- i. the animal or plant is in danger of extinction in Great Britain or likely to become so endangered unless conservation measures are taken;
- ii. for the purpose of complying with an international obligation.

Scheduling is considered to be particularly appropriate where there is a need:

- iii. to protect an animal or plant species from direct human pressure such as persecution, collection or trade;
- iv. to protect elements of habitat essential for the survival of an endangered species.

A species is to be recommended for scheduling if one of the eligibility criteria in each of four sections (A-D) is met. The first three criteria outline eligibility in terms of history, biogeography and taxonomy of a taxon and the final section (D) outlines the endangerment criteria. The first section (A) states that only native (including re-established) taxa are considered by the act and outlines the exceptions by which introduced species may be considered for protection. Taxa can only be considered for protection if they are established in the wild in Great Britain, or vagrant and requiring international protection or extinct as a breeding population and either in the process of re-establishment or could become re-established naturally (Section B). Only well taxonomically authenticated species are considered for protection. Sub-specific taxa are considered provided they are clearly recognisable i.e., morphologically, geographically or ecologically distinct (Section C).

Section D outlines 5 criteria that may indicate that a taxon is, or may become endangered if it is:

- i. listed as threatened in a British Red Data Book or IUCN criteria;
- ii. present at a single locality;
- iii. confined to a particularly threatened habitat, which, is being or is likely to be reduced;
- iv. rapidly declining;
- v. endangered or likely to become endangered through exploitation or collection (more details in Table 3.9).

The decline indicator (iv) refers to at least a 50% decline in the last 20 years (observed, inferred or suspected), or 50% decline projected in the near future in population, number of localities occupied or range. The decline must transcend normal fluctuations.

Table 3.9. Eligibility criteria for scheduling species under the Wildlife and Countryside Act 1981 Great Britain..

For a species to be recommended for scheduling one of the eligibility criteria in each of the sections A to D below should be met:	
A	Generally only native (including re-established) taxa are to be considered. Taxa introduced or thought to be introduced to Great Britain by man could be considered exceptionally, with the following provisos: the organism is endangered or extinct in its native range, and preferably the natural range reaches the north west coast of Europe (which is explicitly defined); for marine taxa, the distribution includes the north west Atlantic area, and provided that information suggests that the organisms is unlikely to have an adverse impact on important native species or ecosystems
B	The taxon must be either: established in the wild in Great Britain or occur as a vagrant in Great Britain and require international protection or be believed extinct in Great Britain as a breeding species, but be in the process of re-establishment or be believe extinct in Great Britain, but with the possibility that it could become re-established naturally.
C	The taxonomic status of the organisms must be well authenticated. Taxa below the species level could be considered, providing they are: clearly recognisable (i.e., morphologically distinct), and geographically or ecologically distinct.
D	The taxon must be endangered in Great Britain or likely to become so unless conservation measures are taken, and/or be subject to and international obligation [1] for protection. One or more of the following may indicate that a taxon is or may become endangered: it is included in a Joint Nature Conservancy Committee-approved Red Data Book as <i>Extinct</i> , <i>Endangered</i> or <i>Vulnerable</i> (or, in Red Lists drawn up using the recently revised IUCN criteria, as <i>Extinct in the Wild</i> , <i>Critically Endangered</i> , <i>Endangered</i> or <i>Vulnerable</i>); it has been well searched for but is known from only a single locality; it is confined to a particularly threatened habitat. The extent or quality of the habitat is being significantly reduced, thus threatening the survival of the organism; it is rapidly declining in population, number of localities occupied or range. Indicative would be at least 50% decline [1] in the last 20 years, or a decline of at least 50% [1] to be likely in the near future. The decline must transcend normal fluctuations; it is endangered, or likely to become endangered through being targeted for exploitation of killing for commercial reasons and/or through being particularly attractive to collectors International obligations apply to a taxon which is: naturally resident and listed on Appendices I, II, III of the Bern Convention; Annexes II, IV, V of the EC Habitats and Species Directive; Appendix I of the Bonn Convention (unless derogations are in force) endemic to Great Britain and included in a JNCC-approved British Red List

Footnotes

1. Observed, estimated inferred or suspected.

3.1.4.2 Endangered Species Act 1973 USA (<http://endangered.fws.gov>)

Aims and geographical scope

This law was enacted to counteract the rates of extinctions faced by fish wildlife and plant species. The Endangered Species Act (ESA) aims to protect endangered and threatened species and the ecosystems and to implement recovery.

Legislative basis and links

The ESA is a complex but flexible law administered by the Interior departments U.S. Fish and Wildlife Service (FWS) and the Commerce departments National Marine Fisheries Service (NMFS). The FWS is responsible for terrestrial and freshwater organisms while NMFS is responsible for marine species. A species is protected by the ESA only after it has been added to the Federal lists: The List of Endangered and Threatened Wildlife and the List of Endangered and Threatened Plants.

A listed species and its habitat are provided with considerable protection from a wide variety of factors at a number of governmental levels with a variety of methods. These include protection from being jeopardized by Federal activities, restrictions on take and trafficking, development and implementation of recovery plans, authorisation to purchase important habitat and Federal aid to State and Commonwealth conservation departments with cooperative species agreements (Nicholopoulos, 1999).

The process

In addition to the listing species and subspecies, the ESA allows the listing of 'Distinct Population Segments' of vertebrates which is based on the concept of an Evolutionary Significant Unit (ESU) (Waples, 1995). A stock must satisfy two criteria to be considered an ESU: it must be substantially reproductively isolated from other conspecific population units, and it must represent an important component in the evolutionary legacy of the species.

There are two methods of listing a species under the ESA; by the petition process or through the candidate assessment process. Any interested person may petition the Secretary of the Interior to add or remove species from the Endangered or Threatened lists. Alternatively FWS or NMFS biologists can identify species through the 'candidate listing process'. Both processes require published findings.

In the case of a petition a 'status review' is conducted within one year if there is substantial information that the petitioned listing *may be* warranted. After a 'status review' there are three possible outcomes: 'Not warranted', 'Warranted but precluded' or 'Listing is warranted'. If listing is warranted a 'notice of review' is published in the Federal Register (http://www.access.gpo.gov/su_docs/aces/aces140.html) and the species is referred to as a 'candidate'. After the proposed ruling is published in the Federal Register any interested person has 60-day comment period to provide additional information or submit statements at public hearings.

Within one year of a listing proposal there are three courses of actions:

- 1) a final listing rule is published in the Federal register, which becomes effective 30 days after publication; or
- 2) the proposal is withdrawn because the available biological information does not support the listing; or
- 3) the proposal is extended for an additional six months, because there is substantial disagreement concerning the biological appropriateness of the listing. After this time a decision is made based on the best available scientific information.

The status of listed species is reviewed every five years.

Listing criteria

A priority system is used to direct listing effort because of large number of candidates and the time required to list a species. The highest criterion is degree or magnitude of threat, followed by the immediacy of threat and finally the taxonomic distinctness of the species (monotypic genus > species > subspecies, variety or vertebrate population) (Table 3.10). Preference is given to popular species or 'higher life forms'.

A number of factors determine whether a species should be listed as endangered or threatened including:

- the present or threatened destruction, modification, or curtailment of the species' habitat or range;
- overutilisation for commercial, recreational, scientific, or educational purposes;
- disease or predation;
- the inadequacy of existing regulatory mechanisms; and
- other natural or manmade factors affecting the species continued existence.

Table 3.10. Terms and definitions of the Endangered Species Act (US).

Term	Definition
Endangered	An endangered species is one that is in danger of extinction throughout all or a significant proportion of its range
Threatened	A threatened species is one that is likely to become endangered in the foreseeable future
Species	Species is broadly defined to include subspecies and distinct populations segments (for vertebrates)
Not warranted	Data do not support need to list
Warranted but precluded	Data support the need to list but other species are of higher priority. In this case an annual re-evaluation is required until a species until either listing is warranted or not warranted
Listing is warranted	Data support the need to list

3.1.4.3 American Fisheries Society

Aims and geographic scope

The American Fisheries Society (AFS) has pursued an initiative to identify marine fish stocks that may be at risk of extinction in North America (Musick *et al.*, 2000). The society has produced a list of fish stocks at risk of extinction for North America, including the coastal waters of Mexico, the continental United States, Alaska and Canada. The among others the main objective of this listing scheme, among others, was to identify stocks at a sufficiently early stage to avoid listing as *threatened* or *endangered* according to the Endangered Species Act 1973 (USA).

Legislative basis and links

The AFS criteria have no legislative basis and like IUCN criteria are a surveillance or guidance tool only. This list highlights species before they become eligible for listing under national endangered species acts.

The listing process

Comprehensive listing of all marine, estuarine and diadromous fishes (excluding Pacific salmonids) has occurred only once (Musick *et al.*, 2000). The compilation of the list involved the use of documentation ranging from published literature, agency reports, and stock assessments and queries of regional scientists and species-group specialists. It is hoped that the list will maintained and updated online.

Listing criteria

Species are selected based on a two-step process that explicitly recognises the intrinsic productivity of species and the rate of population decline. The minimum unit considered by the AFS is the distinct population segment (DPS) of the Endangered Species Act (USA). First, each species (or DPS) is assigned to one of four categories of productivity. While the primary measure of productivity is the intrinsic rate of population increase (r), alternatively growth rate, fecundity, maximum age and age at maturity can be used (Table 3.11). Species are ranked according to the lowest productivity ranking based on whatever data are available (Musick, 1999). Second, the decline of the species is assessed according to thresholds, which vary according to productivity categories (Table 3.11, 3.12). If the population decline were greater or equal to the thresholds, then a species would initially be listed as *vulnerable*. Species faced with additional risk factors may be upgraded to *threatened* or *endangered* based on expert evaluations. Other risk factors include rarity (whether a

rare species is naturally rare or simply cryptic or elusive), small geographic range or endemism, and specialised habitat requirements.

Table 3.11. Potential values for biological parameters for four productivity groups, and their decline thresholds. Adapted from Musick (1999).

Parameter	Productivity			
	Very low	Low	Medium	High
r (yr^{-1})	<0.05	0.05–0.15	0.16–0.50	>0.50
K (yr^{-1})	<0.05	0.05–0.15	0.16–0.30	>0.30
Fec (yr^{-1})[1]	<10 ¹	10 ¹ –10 ²	10 ² –10 ³	>10 ⁴
T_{mat} (yrs)	>10	5–10	2–4	<1
T_{max} (yrs)	>30	11–30	4–10	1–3
Decline threshold [2]	0.70	0.85	0.95	0.99

Foot notes

[1] Fecundity is measured as newborn offspring or newly laid or spawned eggs. [2] Decline is defined using a similar approach to the IUCN or COSEWIC lists, i.e., over 10 years or 3 generations.

Table 3.12. North American marine fish at risk according to American Fisheries Society criteria. Those species for which certain stocks only are considered at risk are indicated (*).

Family	Species
Rhincodontidae	<i>Rhincodon typus</i>
Carchariidae	<i>Carcharias taurus</i>
Cetorhinidae	<i>Cetorhinus maximus</i>
Lamnidae	<i>Carcharodon carcharias</i>
Carcharhinidae	<i>Carcharhinus obscurus</i>
	<i>Carcharhinus signatus</i>
Pristidae	<i>Pristis pectinata</i>
	<i>Pristis pristis</i>
Rajidae	<i>Raja laevis</i>
Acipenseridae	<i>Acipenser brevirostrum</i>
	<i>Acipenser medirostris</i>
	<i>Acipenser oxyrinchus</i>
	<i>Acipenser transmontanus</i>
Clupeidae	<i>Alosa alabamiae</i>
	<i>Clupea harengus</i> *
	<i>Clupea pallasii</i>
Osmeridae	<i>Hypomesus transpacificus</i>
	<i>Spirinchus thaleichthyes</i>
	<i>Thaleichthys pacificus</i>
Salmonidae	<i>Oncorhynchus tshawytscha</i>
	<i>Oncorhynchus nerka</i>
	<i>Salmo salar</i>
Gadidae	<i>Melanogrammus aeglefinus</i> *
	<i>Gadus morhua</i>
	<i>Gadus macrocephalus</i>
	<i>Theragra chalcogramma</i>
Merlucciidae	<i>Merluccius productus</i>
Aplocheilidae	<i>Rivulus marmoratus</i>
Cyprinodontidae	<i>Fundulus jenkinsii</i>
Atherinidae	<i>Menidia menidia</i>
Syngnathidae	<i>Microphidia brachyurus</i>
Scorpaenidae	<i>Sebastes</i> spp
Percichthyidae	<i>Stereolepis gigas</i>
Serranidae	<i>Epinephalus</i> spp.
	<i>Mycteroperca</i> spp.
	<i>Parasphyraenops incisus</i>
	<i>Hypoplectrus gemma</i>
Sciaenidae	<i>Bairdiella sanctaeluciae</i> ,
	<i>Totoalba (Cynoscion) macdonaldi</i>
Stichaeidae	<i>Acantholumpenus mackayi</i>
Anarhichadidae	<i>Anarhichas orientalis</i>
Labrisomidae	<i>Starksia starcki</i>
Eleotrididae	<i>Gobiomorus dormitor</i>
Gobiidae	<i>Awaous tajasica</i>
	<i>Awaous stamineus</i>
	<i>Eucyclogobius newberryi</i>
	<i>Gobionellus pseudofasciatus</i>
	<i>Lentipes concolor</i>
	<i>Sicydium stimpsoni</i>
Acanthuridae	<i>Acanthurus randalli</i>
Scombridae	<i>Scomberomorus concolor</i>
	<i>Thunnus thynnus</i>
Pleuronectidae	<i>Hippoglossus hippoglossus</i> *

3.1.4.4 Committee on the Status of Endangered Wildlife in Canada (COSEWIC) <http://www.cosewic.gc.ca>

Aims and geographic scope

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC), is a centralised, science-based body which is independent of government with the aim of designating wild species in danger of disappearance.

Legislative basis and links

The Species at Risk Act (SARA) 2001 gives COSEWIC a mandated responsibility for producing a Canadian endangered species list. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) was established in 1978, and consists of 20 members representing each of the four federal agencies, and each provincial and territorial government wildlife agency, three non-government scientists and the chairs of its eight Species Specialist Subcommittees (SSCs).

The process

COSEWIC has Species Specialist Sub-committees for both marine and freshwater fish. COSEWIC assess status reports for candidate species, with these reports describing, for example, trends in the distribution range, population size and habitat of the species, anthropogenic threats and significance of the species (Table 3.13). Once the status report is assessed, species are allocated to the appropriate risk category (Table 3.14). The committee can consider species if a suitable status report is received from a third party.

Listing criteria

Like the process in the USA, COSEWIC operates a 'candidate list' that includes species that have not yet been assessed by COSEWIC, but are suspected of being at some risk of extinction or extirpation. Species are assigned a low, medium or high priority on the candidate list. COSEWIC will commission status reports for candidate species as time and resources allow. Status reports contains the best-available information on the basic biology of a species, as well as information on population sizes and distribution in Canada, trends in population sizes, and habitat availability. COSEWIC uses the status report as a basis for discussion and to assign status to a species. COSEWIC uses a criteria-based status assessment process modelled on that used by the World Conservation Union (IUCN) (Table 3.4).

The *Endangered* and *Threatened* criteria are the same as the *Endangered* and *Vulnerable* criteria of the IUCN Red List, however the declining population threshold (A) for threatened species is lower in the COSEWIC listing (20%) than in the IUCN listing (30%).

Table 3.13. Terms and risk categories of Committee on the Status of Endangered Wildlife in Canada (COSEWIC)

Risk category	Definition
Extinct	A species [1] that no longer exists
Extirpated	A species that no longer exists in the wild in Canada, but occurring elsewhere
Endangered	A species facing imminent extirpation or extinction
Threatened	A species that is likely to become endangered if limiting factors are not reversed
Special Concern	A species [is] of special concern because of characteristics that make it particularly sensitive to human activities
Not at Risk	A species that has been evaluated and found to be not at risk
Data Deficient	A species for which there is insufficient scientific information

[1] A species is defined as: any indigenous species, subspecies, variety, or geographically or genetically distinct population of wild fauna and flora.

Table 3.14. The status of marine fishes listed by COSEWIC (excluding species listed as *Not at risk* and *Data deficient*).

Family	Species	Status
Salmonidae	Atlantic salmon <i>Salmo salar</i> ¹	Endangered
	Coho salmon <i>Oncorhynchus kisutch</i> ²	Endangered
Gadidae	Atlantic cod <i>Gadus morhua</i>	Special Concern
Scorpaenidae	Bocaccio <i>Sebastes paucispinis</i>	Threatened
Stichaeidae	Pighead prickleback <i>Acantholumpenus mackayi</i>	Special Concern
Anarhichadidae	Northern wolffish <i>Anarhichas denticulatus</i>	Threatened
	Atlantic wolffish <i>Anarhichas lupus</i>	Special Concern
	Spotted wolffish <i>Anarhichas minor</i>	Threatened

¹: Inner Bay of Fundy populations; ²: Interior Fraser population

3.1.5 Evaluation of the Texel-Faial criteria

The Texel-Faial criteria are used by OSPAR to assess, ‘which species and habitats need to be protected and those human activities that are likely to have an actual or potential adverse affect on these species and habitats or on ecological processes’ within the OSPAR area. Six criteria and associated guidelines are used for the identification of species and habitats in need of protection by OSPAR: global importance, regional importance, rarity, sensitivity, keystone species and decline (Table 3.5).

The mechanisms for applying the criteria may not be standardized at the present time, as some of the criteria are only outlined briefly (OSPAR, 2000, 2003), the guidelines as to their application are not explicit, there is the possibility of some redundancy among the first four criteria, and most of the criteria are not quantitative.

The longer standing and more widely used criteria for assessing the status of species (e.g., IUCN Red List) are now devised in an explicit and well-documented manner (Mace 1995; IUCN, 2004). The need for clear and unambiguous criteria and guidelines requires the appropriate supporting documentation, which provide considerably more detail than the current text outlining the Texel-Faial criteria. For example, the IUCN Red List criteria are outlined in a 24-page document and the guidelines for applying the criteria run to 50 pages. The original IUCN criteria were developed for terrestrial taxa and much emphasis has been given to improving such criteria so that they are more applicable to marine species (see caveat in Table 3.4).

The key utility of an indicator or set of criteria is wide understanding and acceptance by user groups. This can be achieved by devising criteria using a consultation process that are then peer-reviewed by all user groups. It has not been made explicit whether this has been the case for the development of the Texel-Faial criteria.

The first three criteria combine two important metrics - abundance and geographic range size - in non-mutually exclusive ways. This is done in a manner that could be more precise in various cases. For example ‘a high proportion of a species’ as used in the first criterion could be interpreted in terms of numerical abundance or as geographic range. A case could be envisaged where a species with a small population size and found only in a small number of places within the OSPAR area would meet criterion 1, 2 and 3. It is not clear from the criteria whether a species meeting more of the criteria would be afforded a higher priority. The configuration of the first three criteria would preclude such an approach. Various schemes of applying the criteria were suggested (OSPAR, 2000), although an agreed standardized approach is currently lacking.

Qualitative criteria offer the advantage of allowing the examination of species for which data are limited. However, criteria should be as quantitative as possible to facilitate the application of the criteria, and to allow objective decision-making and peer-review. Absence of quantitative thresholds means that decision-making may result in conflict between two different paradigms or schools of thought, e.g., between those supporting resource use values and those supporting conservation or biodiversity values (Mace and Hudson 1999).

“The utility of the Texel-Faial criteria can be examined using the *hits*, *misses* and *false alarms* framework derived from signal detection theory (Rice 2003). A good indicator or criterion will only provide *hits* and minimise the rate of *misses* and *false alarms*. There are two types of *hit*: a ‘true positive’ and a ‘true negative’. In this context a ‘true positive’ would be where the species is listed under the selection criterion and genuinely requires protection, and a ‘true negative’ is where a species not requiring protection also did not meet the criterion (Table 3.15). A *false alarm* is where a species meets the criterion yet does not require protection, and a *miss* is where a species does not meet the criterion yet requires protection.

Table 3.15. The application of signal detection theory to assess the utility of a criterion for selecting species requiring protection.

	Species meets criterion	Species does not meet criterion
Species requires protection	Hit (true positive)	Miss
Species does not require protection	False alarm	Hit (true negative)

The attributes assessed by criterion 1 (Global importance) are not necessarily related to the need for protection. There are a number of species with biogeographical distributions that fall largely within the OSPAR region, which are not in need of protection (e.g., dab *Limanda limanda*). Global importance may have more relevance for management and monitoring issues, as opposed to the identification of declining species, and its use in the application of the Texel-Faial criteria could be better clarified, including its relation to other criteria.

The second criterion (Regional importance) is particularly difficult to interpret. We have interpreted this criterion as aiming to highlight those species with relatively small geographic range sizes, where the bulk of the global range is contained within the OSPAR area and is restricted to a small number of regions or specific habitats within the OSPAR area. It is known that species with small geographic range sizes or are endemic to a few locations may be at a higher risk of extinction. This criterion may also generate ‘false alarms’ by including species restricted to particular habitats but are locally abundant and not in need of protection, and further clarification of the weighting given to this criterion is required.

The third criterion (Rarity) suggests that, for mobile species such as fish, that rarity is dependent on the total population size. This is inconsistent with known theory on rarity, which is based on geographic range size, habitat specificity and population size (Rabinowitz 1981; ICES, 2003a). It should be borne in mind that for many species of fish, our perception of rarity might be biased by the available data from sampling programmes. Such sampling programmes are restricted in space and time, are largely not comparably among regions and only effectively sample a restricted range of the fish assemblage. Taxonomically problematic groups could also lead to misconceptions regarding rarity. Rarity can also be a natural occurrence, and those species that are naturally rare tend to have life-history strategies that reflect this (ICES, 2003a). Hence, the assessment of rarity ought to be viewed in context of what data are available (i.e., are the data sufficiently robust to state that a species is rare), other criteria (e.g., is the rationale for rarity reiterating the rationale for local abundance) and threat (i.e., are there anthropogenic activities that are affecting the species). Population size of the species/stock in question is obviously an integral part of rarity, although quantifying this may be problematic for many species of marine fish, as our current knowledge of total population sizes are poor, as are estimates of the genetically effective population size (i.e., the population required to maintain the genetic diversity of the species/stock) and the minimum viable population size (i.e., the smallest viable population that will persist for a specified length of time and with a specified level of certainty).

The fourth criterion (Sensitivity) requires extensive guidelines for implementation. This criterion may not initially be easy to apply quantitatively because of the lack of theory for determining life history characteristics or otherwise which might be used to determine thresholds beyond which a species is defined as sensitive, although it could be applied qualitatively (e.g., low rate of population increase). This criterion could potentially also be used to highlight those species that occur in threatened habitats.

The fifth criterion (Keystone species) could be loosely applied to a variety of top predators that have a disproportionate influence on other food web components. The key problem with this metric is that the degree to which a species exerts a keystone role cannot easily be measured; therefore it would be difficult to achieve any consensus. There is no direct evidence of any fish acting as a keystone species, and there are fundamental problems with detecting a keystone species, because of the potential non-linearity between predators and prey (May, 1977; Dulvy *et al.* 2004). Hence, this criterion may not be appropriate for fishes.

The sixth criterion (Decline) is one of the more important of the criteria; it is relatively well underpinned by theory and can be measured using survey and other quantitative data for some species. It should be stressed, however, that current surveys may not be appropriate for many species; declines in commercial fish stocks ought to be viewed in the context of the management regimes, and, for many species, climate and other environmental factors affect relative abundance. Misinterpretation of data sets could give an erroneous impression of decline rates (e.g., if data are examined from the year of a strong recruitment event). Hence, quantitative decline rates over appropriate temporal and spatial scales should be used whenever possible in order to utilise this metric and to minimise the likelihood of ‘false alarms’.

At present the Texel-Faial criteria need further development and more explicit guidelines as to how they should be applied, as in their current form it is likely that most of the criteria may produce ‘false alarms’, which is undesirable. Improvements to the Texel-Faial criteria ought to be considered, with issues that need to be addressed including:

- OSPAR should adopt a standardised scheme for applying the criteria and for prioritising threatened and declining species, following peer-review.
- The production and peer-review of documentation for guidelines for the application of the criteria
- The criteria need to be based on current scientific opinion which argues that metrics such as population size / decline rates, geographic range size / endemism and ecological specialisation are the key correlates of threat (Musick 1999; FAO 2000; Reynolds *et al.* 2002; Dulvy *et al.* 2003).
- It should be noted that it is difficult and potentially subjective to define ecological specialism.
- There is potential redundancy between some of the criteria regarding range size, and the use of alternative criteria based on abundance and geographic range could be considered.
- Quantitative decline rates over temporal and spatial scales should be appropriately related to generation times, or a suitable proxy, for the species assessed.
- For those species where discrete and well-defined stocks are known, should the criteria be applied to the species as a whole, or to individual stocks?
- Allocating species listed as “threatened and declining” to more specific criteria (e.g., endangered, threatened etc.), comparable to criteria listed in Section 3.1, would assist in highlighting those species of greatest concern.

3.2 Abundance-range size relationships

3.2.1 Introduction

Abundance-occupancy relationships have been well studied for many terrestrial taxa, in particular British birds (Gaston, 1996a), however, they have not been as extensively explored for fishes or other marine taxa. It is a general ecological observation that relatively abundant species are also geographically widespread, whereas species that are rare typically have a narrower geographic distribution (Gaston, 1996a). This pattern is apparent within species and among groups of species, and abundance-occupancy distributions are regarded as an almost universal feature of animal assemblages (Gaston and Blackburn 2000). The terms occupancy, distribution and range size generally reflect the geographic spread of the organisms in question. The use of these different terms reflects the type of data used. Range size or distributional area are used when latitudinal or area units (km²) are used, whereas the terms ‘occupancy’ and ‘incidence’ generally represent the proportion of an area, usually within a grid system, where species are recorded as present. There are a number of persistent properties of abundance-occupancy relationships. They appear to be driven by changes in maximum abundance; the relationship is consistent between different habitats, and stable from one season to another and from one year to the next, at least in British birds (Gaston and Blackburn 2000).

Abundance-occupancy relationships have implications for inventories of biodiversity, population monitoring, fisheries, and conservation (Gaston, 1999). It is often suggested that those species with restricted distributions and low abundance are at greater risk of over-exploitation or, in extreme circumstances, extinction (Gaston, 1999). For fisheries in particular there are a number of aspects relevant to abundance-occupancy relationships, such as the degree to which spill-over effects may occur around closed areas, distribution shifts in response to reduced abundance, elevated catchability as abundance and occupancy are reduced, identification of core habitat / hotspots, minimum viable population sizes, and scaling from local to regional dynamics.

Significant positive interspecific relationships have been described between abundance and occupancy of estuarine invertebrates in the British Isles (Foggo *et al.* 2003), certain freshwater fishes (Pyrone, 1999) and demersal marine fishes off southern Africa (Macpherson 1989). Among flatfishes (reviewed by Gibson 1997), no consistent pattern between abundance and range size have emerged, with some species increasing their geographic range with increasing abundance, while others show either no effect or even decreased distributional area with increasing abundance.

Intraspecific relationships between abundance and distributional area (or occupancy) have been described for several commercially important marine fish species at the population level and within populations for specific ages (Winters and Wheeler 1985, Myers and Stokes 1989, Crecco and Overholtz 1990, MacCall 1990, Rose and Leggett 1991, Swain and Wade 1993, Marshall and Frank 1994, Swain and Sinclair 1994, Hutchings 1996, Swain and Morin 1996, Brodie *et al.*, 1998, Overholtz 2002). It has been shown that distributional area declines with decreasing total abundance, resulting in negative exponential relationships between targeted catchability and distributional area driven by the non-random searching behaviour of fishers (Paloheimo and Dickie 1964, Arreguín-Sánchez 1996). This implies a greater disproportional vulnerability of populations at low population levels to both fishing and predation impacts (predator-pit). If the distribution of a species becomes restricted to core habitat ("hyper aggregation") followed by re-distribution of fishing effort into these areas, it can make fisheries catch rates insensitive to changes in population size and may lead to a false impression of the status of the stock as well as rendering it more vulnerable to collapse. Such events have been documented for Canadian cod (Rose and Kulka, 1999) and sandeels (Wright and Bailey, 1993).

The observed positive or negative intra-specific relationships for several marine fish has focused on density-dependent habitat selection (DDHS) as a mechanism for the coupling of abundance and distribution, after ideal free distribution theory (Fretwell and Lucas 1970) was applied to marine fish populations by MacCall (1990). This approach forms a link between habitat suitability and changes in geographical distribution. Although this mechanism has found some support (Marshall and Frank 1995, Swain and Wade 1993, Swain and Sinclair 1994, Myers and Stokes 1989), it is one of the several hypotheses which has been put forth to explain intraspecific/interspecific patterns in abundance-occupancy (Hanski *et al.* 1993, Gaston *et al.* 1997, Watkinson *et al.* 2003).

Despite the paucity of studies examining the relationships between distribution (or occupancy) and relative abundance of marine fishes in European seas, such analyses may provide a useful method of highlighting species or stocks of concern. There are, however, many biotic and abiotic factors that will determine, or restrict, the overall distribution of a species, including habitat availability, migratory and dispersal capabilities, predator-prey interactions and environmental conditions (Gaston, 1996b). Hence, some care is needed in interpreting abundance-range size relationships.

The macroecological comparative approach has generated a number of repeated patterns, and has provided some insight into the underlying mechanisms (Gaston and Blackburn 2000). However, recent theoretical developments have provided considerable insight into the processes underlying abundance – occupancy relationships. Three mechanisms have been explored to date: vital rates (i.e., population growth, mortality, survival), metapopulation processes and density-dependent habitat selection (Watkinson *et al.* 2003; Freckleton *et al.* in press). Analytical models have been developed based on linking the amount of suitable habitat to vital rates such as the population growth rate (Holt *et al.* 1997; Freckleton *et al.* in press). Figure 3.2.1 shows an informal model of how variation in a vital rate, the finite rate of population increase (B) may relate to changes in total population abundance (N) over two types of habitat: continuously distributed habitat (panels a and b) and patchily distributed habitat (panels c and d). Increasing the finite rate of population increase increases the potential range of species over both habitats, but this is less pronounced in patchy habitats (panel a and c). Increasing the finite rate of population increase increases both the local population size and the incidence.

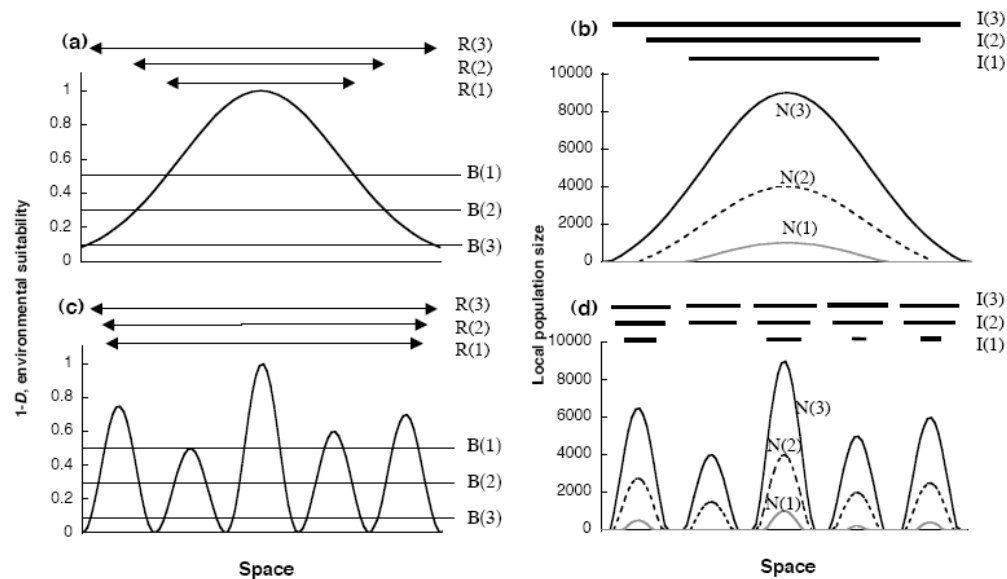


Figure 3.2.1. Schematic models for range-occupancy-abundance relationships. (a) and (b) follow patterns of range and occupancy expansion in an area of continuous habitat. (a) The curve represents habitat suitability as is defined by $1-D$ where D measures mortality. As the finite rate of increase (B) is changed from a low value, $B(1)$, through to a high value, $B(3)$, the potential range (R) increases as shown. (b) Habitat is continuous, hence incidence (I) is the same as the total range. Increasing the finite rate of increase from $B(1)$ to $B(3)$ leads to an increase in population size ($N(1)$ to $N(3)$), since increasing B increases the difference between production of offspring and mortality. (c) and (d) show a second model in which habitat is distributed patchily. (c) Habitat suitability is patchy rather than continuous as in (a). This means that as B is varied the impacts on geographic range size (R) are rather small. (d) By contrast there is an effect of varying B on the incidence of species within the habitat (I), as well as local population density (N). Both models predict that the effect of increasing B on incidence should be saturating.

Both models predict the existence of an asymptotic relationship between occupancy and local population size and this is shown in Figure 3.2.2a, for a range of parameter values. At low population densities occupancy (incidence) is proportional to local population size, whereas at high population densities occupancy is independent of local population size. This has important implications for the relationship between local abundance and total abundance (local abundance \times occupancy). When local abundance is low total abundance is proportional to N^2 but when N is large occupancy changes independently of N and total abundance scales to N (Figure 3.2.2b). This is shown in Figure 3.2.2c the slope of the relationship between total and local abundance declines from 2 at low local abundance to 1 at high population sizes.

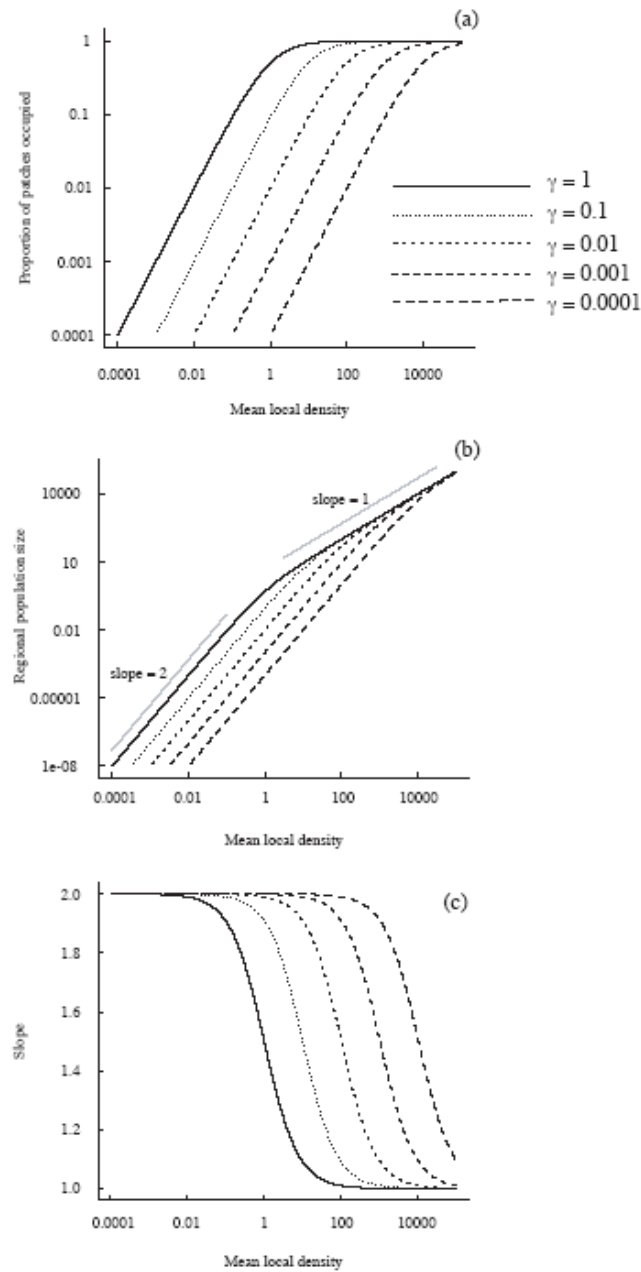


Figure 3.2.2. Abundance-occupancy relationships in a habitat-filling model. (a) Relationship between occupancy and average local population density; (b) relationship between regional population size (defined as the total population size divided by the maximum number of patches) and local population density on a double logarithmic scale; (c) slope of the relationship in (b).

Metapopulation dynamics consider the proportion of suitable habitat patches that are occupied relative to the balance between the extinction rate within patches and the colonization rate of patches (Levins 1970; Levins and Culver 1971). As populations decline to the point where extinction exceeds the colonization rate the total population size declines precipitously resulting in steep scaling at low local densities resulting in initial slopes of the local density – total population size greater than 2 (Figure 3.2.3a,b). Also the existence of metapopulation and rescue effect can be diagnosed by the existence of a minimum local density below which populations do not exist, i.e., the point where the smallest total population size is associated with a large local mean density (Figure 3.2.3c).

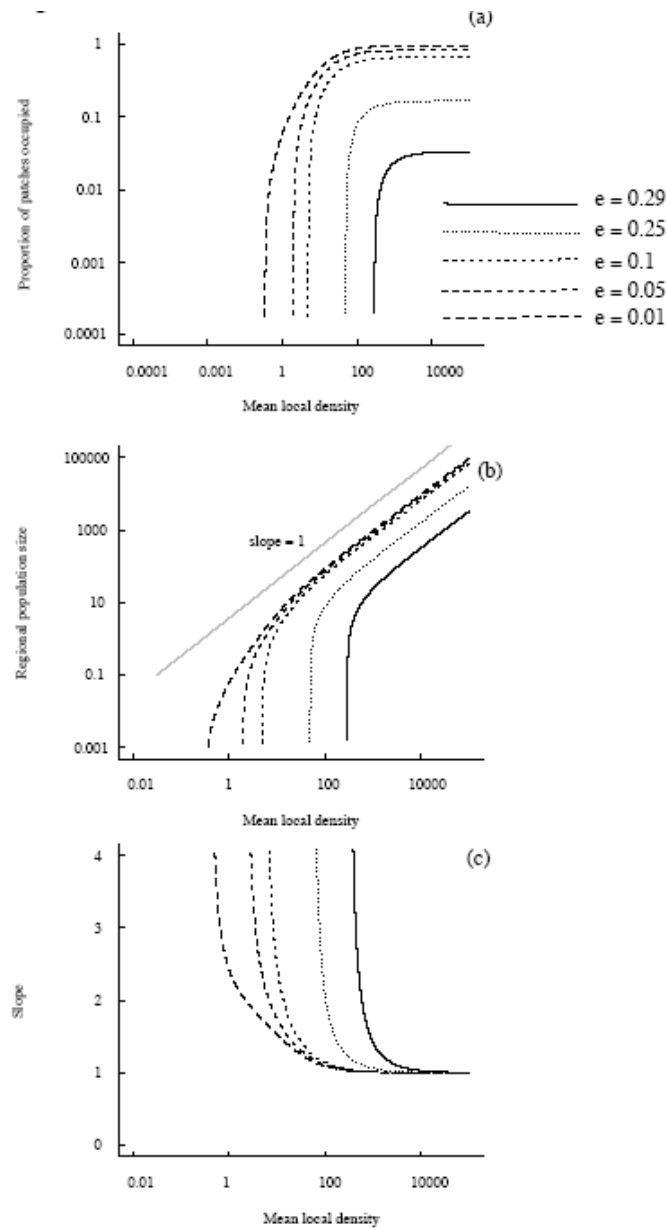


Figure 3.2.3. Abundance-occupancy relationships in the metapopulation model for a fixed colonization rate and varying extinction rates, e . (a) Relationship between occupancy and average local population density; (b) relationship between regional population size (defined as the total population size divided by the maximum number of patches) and local population density on a double logarithmic scale; (c) slope of the relationship in (b).

These simulations show that the nature of regional population dynamics, such as colonization and extinction dynamics, determine the form of the abundance-occupancy relationship. Also the nature of the abundance-occupancy relationship will be informative of the factors that determine large-scale population dynamics. A number of predictions have been made from these habitat-filling models including;

- 1) For species with low colonization rates, there should be no abundance-occupancy relationship;

- 2) For species with high colonisation ability, a simple hyperbolic abundance-occupancy relationship is predicted; and
- 3) Species with moderate colonisation ability will exhibit metapopulation dynamics and a minimum density exists below which populations cannot persist.

This simulation analysis shows that abundance–occupancy relationships are important because they link local and regional population processes and allows the local and regional impacts to be explored simultaneously (Watkinson *et al.* 2003; Freckleton *et al.* in press). Clearly such patterns have implications for fisheries and the ecosystem approach to management. Interspecific differences in abundance-occupancy could be used to highlight those species that are potentially at risk and, if subsequently found to be at risk, those instances where spatial management would be the most effective management regime. In contrast, species with a large range size, which still may be at risk to human activities, may benefit from more general management measures. Intraspecific differences in abundance-range size could be used to examine temporal trends for species or stocks of fish, and groundfish surveys may be appropriate for selected species.

3.2.2 Case study: Preliminary examination of interspecific marine abundance range-size relationships (CEFAS)

Relationships between occupancy and relative abundance in demersal fish and shellfish were examined for data collected in the Bristol Channel (VIIIf) by 4m beam trawl (1991–2003). During this period, 411 standard tows were made at 32 stations that were fished in most years (those stations that were not fished consistently were excluded from the analysis).

The general relationship between abundance and occupancy is given by the following equation:

$$\text{Log}[p/(1 - p)] = \alpha + \beta \log \mu'$$

Where p = the proportion of catches in which the species was present, and μ' is the average density of the species at occupied sites (Foggo *et al.*, 2003). Aggregated data confirmed that there was a general tendency for the most abundant species to be found at a greater proportion of sites (Figure 3.2.4, $r^2 = 0.62$), although in the case of groundfish surveys, this may be bias by the design (stratification) of the sampling grid. Plots comparing p (untransformed) with μ' gave broadly similar results (Figure 3.2.5, $r^2 = 0.68$).

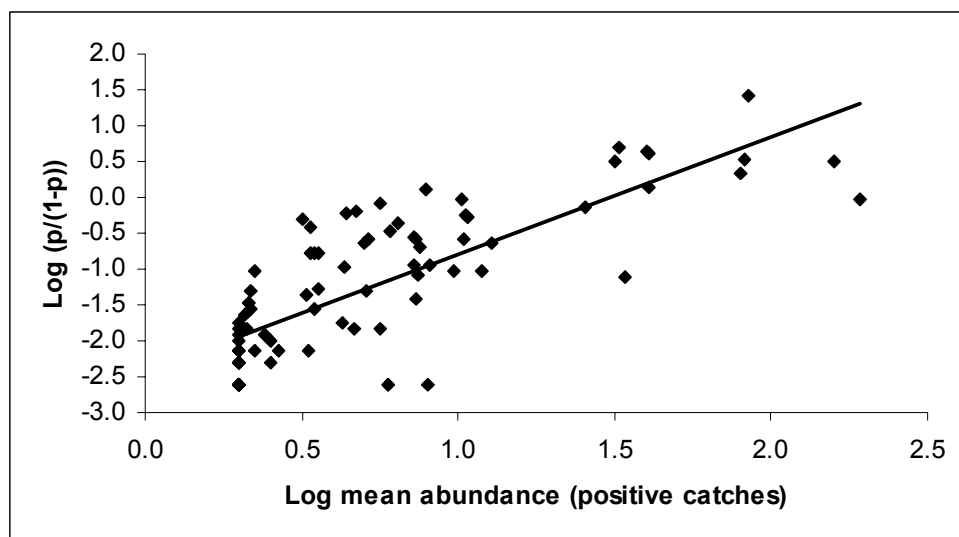


Figure 3.2.4. Relationship between log occupancy and log abundance for fish species in the Bristol Channel.

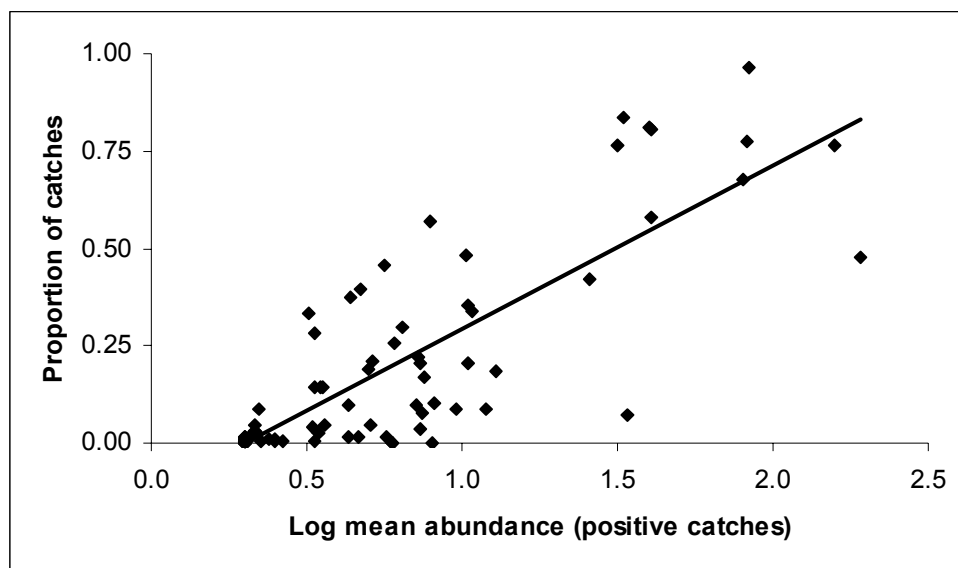


Figure 3.2.5. Relationship between occupancy and log abundance for fish species in the Bristol Channel.

For surveys with a standardised sampling grid, such analyses could be undertaken on a species-specific basis with which to assess temporal trends in the abundance-range size relationship (Figure 3.2.6). Although there was a significant relationship between abundance and occupancy for some species (smalleyed ray, $r^2=0.67$), no significant relationships were observed for other species (e.g., common dragonet and solenette).

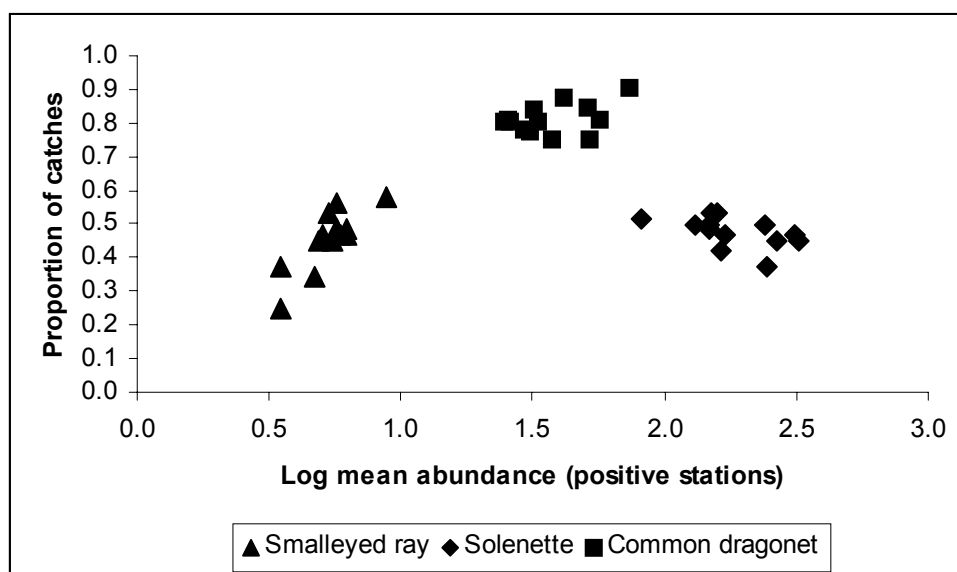


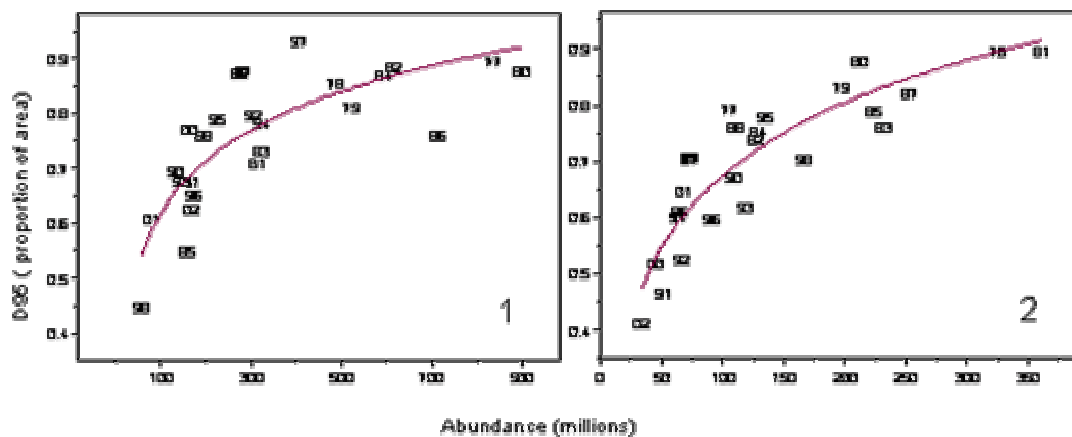
Figure 3.2.6. Relationship between occupancy and log abundance for smalleyed ray *Raja microocellata*, solenette *Buglossideum luteum* and common dragonet *Callionymus lyra* in the Bristol Channel (1991–2003).

3.2.3 Case study: Distribution-abundance relationships for juvenile (ages 1 and 2) North Sea cod (CEFAS)

Along with examining geographic range size relationships for species, another approach is to investigate age-specific patterns within populations. Relative catch density of cod was expressed as mean numbers per standardized tow (per hour) for each ICES rectangle from the English Groundfish Survey. Indices of total abundance for juvenile cod (ages 1 and 2) in the North Sea were obtained from numbers at age in the most recent VPA (ICES, 2003b) for ICES areas IV, VIId and part of IIIa (North Sea, English Channel and Skaggeerak).

To examine the abundance-distribution relationship for age 1 and 2 cod, we measured the area occupied by the population in each year as the sum of ICES rectangles that contained > 95% of the population abundance as estimated from the cumulative distribution of catch densities. This approach was used to allow for general comparison with Swain and Sinclair (1994). The adjustment to the total number of rectangles surveyed in each year for both indices was necessary to account for the fact that the spatial coverage of the survey during 1977–1982 was less than the later part of the time series, and to account for differences in spatial coverage from year to year. Some stations are not sampled every year due to poor weather or equipment failure.

$$D = \beta_0 + \beta_1 \cdot \ln(N)$$



The positive curvilinear relationship between the distribution and abundance of juvenile North Sea cod, as seen above, is consistent with the abundance-distribution patterns observed for Canadian cod stocks (Swain and Sinclair 1994, Swain and Wade 1993, Hutchings 1996). One of the mechanisms driving this relationship could be density dependent habitat selection as the observed survey distributions were consistent with an ideal free distribution model when habitat suitability was defined as optimal temperature for growth rate (Blanchard *et al.*, in preparation). However, there are several other possible explanations for these patterns, and consideration of other mechanisms (e.g., other factors influencing habitat suitability, habitat patchiness, vital rates, dispersal and patch extinction), as outlined by Freckleton *et al.* (in press), is required.

- General patterns of increasing geographic range with population size are a widespread ecological phenomenon that extends to animals in the marine environment.
- These patterns can be detected using fisheries-independent survey data. Several approaches can be used depending on the type of data and the form of abundance – occupancy relationships can be determined at both species and population levels.
- The form of the abundance-occupancy relationship has potential use for inferring the processes or mechanisms (e.g., vital rates, density dependent habitat selection, metapopulation dynamics) that are likely to underlie such relationships.

- A contraction of range size as population declines can have serious implications for fisheries management for several reasons. A few of the implications and links with other areas include: higher vulnerability of individuals to fishing and predation at low population size and distribution due to the non-constant negative relationship between targeted catchability and population area; potential for population spill-over from marine protected area; determinants of habitat quality and essential fish habitat.
- Conventional temporal analyses do not directly consider the consequences of spatial distribution. Further work is required to explore the effect of sampling issues on the form of the abundance-occupancy relationship, the processes and mechanisms underlying marine abundance-occupancy patterns and also to explore the utility of abundance-occupancy patterns in understanding management-related issues.
- Abundance-range size relationships need to be viewed in the context of habitat suitability and extent.

3.3 Developing criteria to prioritise fish species for assessment

The use of a candidate list for identifying those species that are nominated to be listed as “threatened and declining” was discussed and it was concluded that a list of species that may be in need of protection could be compiled by an appropriate organisation, and updated regularly.

Such a candidate list could then be reviewed periodically by a suitable organisation, and the list prioritised. Species prioritised should then have detailed status reports submitted, which should assess the current status of the population examined in relation to temporal trends in population size/extent, using defined decline criteria, and existing threats. Such status reports should then be peer-reviewed by a suitably qualified body, and comments forwarded to OSPAR.

In terms of criteria, species requiring protection could be identified using internationally recognised criteria, which have been created in a consultative manner and subject to international peer review. This approach has been taken by a number of international and regional fora such as the Convention on the conservation of European wildlife and natural habitats (Bern Convention), the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) and a number of national fora including the UK Wildlife and Countryside Act and the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). These fora all use criteria similar to the IUCN Red List to identify and document those species most in need of conservation attention. A standardised (and peer-reviewed) approach to the application of the Texel-Faial criteria is required.

A set of suitable criteria for assessing data poor species is required. This would require a non-subjective approach, the development of which could incorporate the use of life-history correlates. Ensuring a rigorous and defensible assessment of such species, in the absence of quantified data, will be challenging.

A system for highlighting those species that fall primarily under the responsibility of OSPAR, as opposed to the remit of other national, regional and international organisations or agreements, could be investigated.

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4 FOOD RATIONS, PREY COMPOSITION AND GASTRIC EVACUATION RATES OF GADOIDS IN THE NORTH SEA

4.1 Introduction

Information about food ration, prey composition and gastric evacuation rate of gadoids in the North Sea is required to improve MSVPA models. The main objective of this section was to do a thorough revision of the estimates of food ration and prey composition of the gadoid MSVPA predators of the North Sea based on the latest improvements in gastric evacuation modelling. The new estimates were compared with the food rations used at present by ICES to estimate predation mortalities of the MSVPA prey fish populations in the North Sea.

Recent research has shown that mean consumption rate over time and population of predatory gadoids may be estimated by:

$$\hat{C} = \overline{\rho S^{0.5}} \quad (1)$$

where S and ρ are total mass and evacuation rate constant, respectively, of the stomach content of each individual sampled in the field (Andersen, 2001). The ICES stomach contents data were generally pooled by predator size. Therefore, only mean values \bar{S} of stomach contents are available. However, the values of $\overline{S^{0.5}}$ and $\bar{S}^{0.5}$ differ unless all values of S are equal. Consumption rates as estimated by:

$$\hat{C} = \bar{\rho} \bar{S}^{0.5} \quad (2)$$

are therefore likely to be biased by the frequency distribution of S . This bias may be remedied by introduction of a correction factor $k = \overline{S^{0.5}} / \bar{S}^{0.5}$ obtained from data on individual stomachs (Andersen, 2001):

$$\hat{C} = \bar{\rho} k \bar{S}^{0.5} \quad (3)$$

The correction factor may be further refined splitting the influence of the S variation into the frequency of empty stomachs ($S = 0$) and the frequency distribution of S from stomachs with food. This way, the information about the frequency of empty stomachs that are available for the pooled ICES data may be used to modulate the value of k by:

$$k = k_F p_{F0.5} \quad (4)$$

where k_F is calculated from S values of the non-empty stomachs of representative sets of individual stomachs, and p_F is the proportion of non-empty stomachs within the relevant pooled set of ICES stomach data (ICES, 2003). Calculations from various North Sea predators have shown that the values of k_F obtained from different data-sets of a predator species are relatively similar whereas the proportion of empty stomachs heavily influences the k estimate (ICES, 2003).

The contents of stomachs sampled in the wild often consist of more prey types. In contrast, laboratory studies on gastric evacuation have almost exclusively considered evacuation rate of meals of one prey type. Traditionally, total food ration has then been estimated either by application of the evacuation rate of a prey type that is a major representative of the stomach contents or by use of one or another kind of mean evacuation rate based on different main categories of prey for which appropriate estimates of the evacuation rate are available (e.g., Daan, 1973; Jones, 1978). Subsequently, consumption of the individual prey type was calculated by its mass fraction of total stomach content. Alternatively, prey specific values of the rate parameter have been used to estimate consumption rates of individual prey types which are then summed to obtain total food ration (dos Santos and Jobling, 1995). However, the rationale for using these different methods is unclear, and except from a test of their food consumption model by dos Santos and Jobling (1995) no attempts have been made to validate the approaches. A better understanding of how different food types in a stomach interact in the process of evacuation is therefore a prerequisite for more reliable estimates of food consumption rate of wild fishes.

A simple, mechanistic gastric evacuation model has been established following a geometric interpretation of the square root model (Andersen and Beyer, *in prep*). Primary and interactive effects of size, energy density and resistance to digestion of individual prey in a stomach were described by the model. Model simulations of results from experiments on gastric evacuation of meals composed of different prey types demonstrated the capability of this model, unlike previously applied model principles, to predict evacuation of mixed meals involving the three above-mentioned prey characteristics. The study also illustrated that estimates of food ration might be severely biased by use of improperly formulated effects and interactions of prey characters, and indicated that the new model is an appropriate candidate model to estimating food rations of wild populations of predatory gadoids.

Because of the interactive effects of the different prey characteristics it is recommended to use the evacuation model to data on the content of each individual stomach estimating food rations in order to avoid the possibility of introducing excessive bias. This was not possible to do here because the stomachs sampled by ICES generally were pooled into predator length groups from each haul. Data on mean stomach content from each individual haul probably give a more realistic picture of prey composition of individual stomachs than do those obtained from averaging over larger geographical areas (ICES squares, round fish areas, and total North Sea). Food rations were, therefore, estimated from data at all levels of averaging to evaluate the effects of data aggregation level.

4.2 Methods

Food rations were estimated by use of information from the international sampling project in 1991 undertaken by ICES and contained in the North Sea stomach database (information about total numbers of hauls and stomachs are listed in Table 4.1). Size measures of individual prey items in the stomachs were not generally included in the database. Therefore, the effects of prey size could not be taken into consideration. Fortunately, this prey character is the least critical unless the size differences are very large (Andersen and Beyer, *in prep*). However, the surface considerations of the concept of Andersen and Beyer (*in prep*) had to be excluded. This reduces the food ration model to the expression for estimated intake of prey type i by

$$\hat{C}_i = \bar{a}_i \bar{\rho}_{i,E} \bar{E}^{-0.86} k \bar{S}^{0.5} = \bar{a}_i \bar{\rho}_{i,E} [(\sum \bar{a}_i \bar{\rho}_{i,E} \bar{E}_i)(\sum \bar{a}_i \bar{\rho}_{i,E})^{-1}]^{-0.86} k \bar{S}^{0.5} \text{ g} \cdot \text{h}^{-1} \quad (5)$$

and estimate of total food ration by

$$\hat{C} = \sum \hat{C}_i \quad \text{g} \cdot \text{h}^{-1} \quad (6)$$

where the mean gastric evacuation rate constant $\bar{\rho}_i$ for prey type i was expanded as a function of means of predator body size and temperature as well as mean energy density of evacuated chyme ($\bar{\rho}_{i,E} = \bar{\rho}_{i,LTE} \bar{L}^{1.44} e^{0.078\bar{T}}$). \bar{L} is mean length (cm) of the gadoid predator, \bar{T} is mean temperature (°C), \bar{E} is energy density (kJ·g⁻¹) of evacuated chyme, \bar{E}_i is mean energy density of prey type i , and \bar{a}_i is mean mass fraction of stomach content constituted by prey type i at the applied level of aggregation of stomach data. The coefficient of variation (CV) for each estimate of total food ration was obtained by boot-strapping.

The prey specific rate constant $\rho_{i,LTE}$ expresses the resistance to digestion of prey type i . The value of this constant may be considered similar for all fish prey (Andersen, 2001). Estimates of $\rho_{i,LTE}$ of fish prey for evacuation in the gadoids whiting, saithe and cod were obtained by Andersen (2001) and for evacuation in haddock by A. P. Robb (Marine Laboratory, Aberdeen, unpublished results). Robust exoskeletons of a variety of different invertebrates are known to increase the resistance to digestion (Bromley, 1994). Evacuation rate constants for relevant groups of these types of prey were obtained from different published as well as unpublished sources.

A general value of 0.85 for k_F [equation (4)] was applied for all predators. It is based on estimates obtained by ICES (2003) as well as unpublished sources.

The quarterly mean energy density of prey type i was estimated from its size class distribution in the stomachs combined with the energy density by prey size class (Pedersen and Hislop 2001; and unpublished data).

Data on the geographical distribution of quarterly mean bottom temperature [estimated by J. -P. Herrmann, Hamburg University, from a 3-D circulation model of the North Sea (Pohlmann 1996)] and abundance of the gadoid predator by age in 1991 [from ICES International Bottom Trawl Survey (IBTS) database, Copenhagen] were combined to estimate mean temperatures at the different aggregation levels of stomach data that was used for the evacuation model.

Quarterly prey type specific and total food rations (g) were estimated as $2190 \text{ h} \times \hat{C}_i \text{ g} \cdot \text{h}^{-1}$ and $2190 \text{ h} \times \sum \hat{C}_i \text{ g} \cdot \text{h}^{-1}$, respectively.

4.3 Results and Discussion

In general, the new estimates of total food ration obtained for the four gadoid predators were not influenced substantially by the level of data aggregation at which the gastric evacuation model was applied (Table 4.2). This is in accordance with the results of Andersen and Beyer (*in prep*), who found that different ways of modelling the effects of prey characteristics on gastric evacuation did only to a limited extent influence the estimate of total food rations.

The new estimates of total food ration differed, however, generally from the old ones used at present by ICES to acquire predation mortality figures for the MSVPA prey fish populations (Table 4.3). The new rations estimated for whiting amounted to between a third and a half of the old food rations only. The new estimates obtained from saithe were on the whole substantially lower than the old figures. They also increased at a faster rate with increasing predator age as did the new estimates obtained from cod. No significant differences were found for most seasons and age groups of haddock. The new whiting and saithe estimates have been validated by thorough bioenergetics studies performed by Andersen, Sand and Jordan (pers. comm.) and Andersen and Riis-Vestergaard (*in press*). Use of the old estimates implied that the fishes needed to swim at maximum sustained swimming speed most of their time, which is highly unrealistic.

The estimates of consumption of individual prey groups/species by the gadoid predators were affected differently by the applied level of data aggregation (Table 4.4). Substantial differences were found in the cases where the prey differed significantly by their energy densities or resistance to digestion. For example, the proportions of energy-rich sprat in food rations consumed by whiting increased at increasing level of applied data aggregation, while the fraction constituted by the prey group 'Other fish' composed of lean fish species decreased. This could be explained by the change in prey composition of stomach content to which the evacuation model was applied when higher levels of data aggregation level was used: On haul level the evacuation rate of the sprat was low because fat, high energy sprat dominated the content of the stomachs from hauls in which it was present. In contrast, the evacuation rate of sprat was significantly increased as estimated, for example, from the content of a 'quarterly mean stomach' representing the entire North Sea because the other prey types decreased the overall energy density of the content. The opposite trend applied to the prey group 'Other fish'. The results represented in Table 4.4 show that the composition of consumed prey is not necessarily the same as the composition found directly from the content of sampled stomachs. It should also be noticed that the differences observed here are absolute minimum values because food rations based on information on individual stomach are not included.

- On the basis of the present results it is recommended to use the new gastric evacuation model rather than the approach employed at present by ICES to estimate food rations.
- It is further suggested to apply the evacuation model to data on individual stomach content in order to estimate food rations or, alternatively, to exploit the lowest aggregation level of stomach data possible in cases where stomach contents have been pooled.
- Finally, it is recommended to examine the consequences of using the new ration estimates in relevant runs of MSVPA/FOR as compared to the results obtained from the old ration estimates with a view to re-evaluating predation mortalities of the MSVPA prey fish populations.

4.4 Relevant ongoing studies

The BECAUSE project

The BECAUSE project (Critical Interactions Between Species and their Implications for a Precautionary Fisheries Management in a Variable Environment – a Modelling Approach), funded by the EC under Framework Programme 6 (Contract 502482), commenced on 1 March 2004 and is scheduled to last three years. BECAUSE will essentially re-examine the potential of various multi-species modelling approaches as operational tools in the implementation of an ecosystem approach to management. At the Intermediate Ministerial Meeting at Bergen in 1997, ministers required that the "ecosystem approach" should involve a "multi-species approach" to management. Thus, interactions between species, between predators and their prey, and between predators competing for a limited prey resource, should be taken into account when managing the exploitation of marine natural resources.

The BECAUSE consortium has identified what they consider to be key predator-prey interactions involving exploited predator species and exploited prey species in several different maritime regions. Each region, the North Sea, Nordic Seas, the Baltic Sea, the Iberian Shelf/Bay of Biscay, and the Mediterranean Sea, will be treated as a separate case study. The interactions to be examined include the predation of predatory fish on sandeels, cod on *Nephrops*, cod on capelin, cod on herring, cod on sprat, hake on prey fish, and two cannibalistic interactions, predation of hake on hake and cod on cod. Each interaction will be examined in the appropriate regions, and in each case, the interaction will be examined within the context of a conceptual food web model. Thus in the case of the main North Sea case study, the predatory fish - sandeel interaction will be examined and modelled within the context of:

- Predatory fish (cod, haddock, whiting) populations that are currently depleted, which if subject to “recovery programmes”, could increase leading to raised predation loadings on sandeels from predatory fish.
- Seabird populations, currently close to the highest population levels recorded over many decades, which may well compete with “recovering” gadoid populations for the sandeel resource.
- Variation in marine mammal populations, some of which are currently at high levels, whilst others are at low levels but which may recover following efforts to reduce bycatch in fisheries, and may compete with “recovering” gadoid populations for the sandeel resource.
- High levels of sandeel removals, between 0.5 and 1 million tonnes, from the North Sea each year by industrial fisheries.
- Variable abundance of sandeel prey, both in time and space, requiring biologically reasonable representation of predator functional responses.
- Variable abundance of alternative prey species, both in time and space, requiring biologically reasonable representation of prey-selection processes.
- Incorporation of recent advances in modelling gastric evacuation rates and so estimating rates of food consumption by fish predators.
- Variable environmental factors (e.g., hydrographic conditions) that influence predator population dynamics and distributions, prey population dynamics and distributions, and availability of prey to predators.
- The potential impact of feedback processes on predator population dynamics, i.e., taking into account the possible consequences of variation in food intake rates on predator growth, body condition, maturation and fecundity.

Results of the analysis of these conceptual food web models will be used to update and improve existing MSVPA models and to parameterise alternative more modern models such as GADGET, BIFROST, SEASTAR, etc. These latter models will be applied to maritime regions other than the one where they were originally developed. The models will be run using historical time series data to examine the implications of incorporating multi-species interaction and environmental forcing for fisheries management. The potential for the models to provide reference points for managers and identify possible ecosystem conflict points will be investigated.

4.5 References

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Table 4.1. Number of hauls (samples) and stomachs by predator species and quarter of the year (1991).

Quarter of the year	Cod		Haddock		Saithe		Whiting	
	Haul (N)	Stomach (N)	Haul (N)	Stomach (N)	Haul (N)	Stomach (N)	Haul (N)	Stomach (N)
1	296	2 182	105	2 442	69	798	190	6 152
2	277	3 174	188	2 768	76	1 191	306	11 330
3	201	2 373	208	4 971	39	402	279	11 543
4	176	1 999	115	2 927	53	858	216	9 406
Total	950	9 728	616	13 108	237	3 249	991	38 431

Table 4.2.a. New estimates of total food ration (g) of cod by age group and quarter of the year (1991), and associated coefficients of variation (CV) obtained by boot-strapping. Gastric evacuation model was applied to stomach content data at different levels of aggregation (haul, ICES square, round fish area, and total North Sea).

		Estimate				CV			
		Sample	Square	Round	Total	Sample	Square	Round	Total
quarter 1	age 1	56	56	58	59	7	7	8	8
	2	486	491	521	531	5	5	5	5
	3	1470	1493	1584	1561	4	4	4	4
	4	3363	3424	3800	3732	4	4	5	6
	5	4611	4707	5251	5222	5	5	6	6
	6	5351	5473	6081	6088	6	6	7	7
2	1	130	131	134	134	4	4	4	5
	2	1076	1091	1133	1134	3	3	3	3
	3	2688	2715	2837	2863	3	3	3	3
	4	5663	5708	5989	6147	4	4	4	4
	5	8707	8785	9247	9509	6	6	6	6
	6	11989	12040	12766	13099	8	9	9	9
3	0	23	23	24	23	19	19	18	16
	1	317	320	331	337	5	5	5	5
	2	1315	1320	1313	1287	4	4	4	4
	3	2466	2480	2409	2292	4	4	4	4
	4	4536	4536	4271	4187	6	6	6	6
	5	5351	5358	4778	4840	9	9	10	9
4	6	8360	8364	8025	8497	19	19	19	18
	0	70	70	72	73	15	15	15	15
	1	466	472	477	483	5	5	5	6
	2	1812	1824	1835	1854	4	4	5	5
	3	3733	3747	3769	3757	5	5	5	6
	4	5741	5755	5824	5777	7	7	7	7
	5	11031	11033	11546	11446	15	15	15	13
	6	14366	14366	15187	15042	20	20	20	18

Table 4.2.b. New estimates of total food ration (g) of haddock by age group and quarter of the year (1991), and associated coefficients of variation (CV) obtained by boot-strapping. Gastric evacuation model was applied to stomach content data at different levels of aggregation (haul, ICES square, round fish area, and total North Sea).

		Estimate				CV			
		Sample	Square	Round	Total	Sample	Square	Round	Total
quarter	age								
1	1	30	30	33	33	6	6	8	8
	2	74	74	76	77	5	5	5	5
	3	122	122	129	130	8	8	7	7
	4	169	169	176	181	10	10	9	8
	5	188	188	196	201	10	10	9	8
	6	287	287	296	307	13	13	12	11
2	1	96	96	98	99	4	4	3	3
	2	196	197	205	206	4	4	3	3
	3	277	280	302	305	4	4	4	4
	4	363	368	402	408	5	5	4	4
	5	337	341	374	380	5	5	4	4
	6	295	297	330	334	4	4	5	5
3	0	15	16	17	17	6	6	7	7
	1	118	119	130	130	8	8	8	8
	2	243	245	260	259	6	6	6	6
	3	386	390	424	427	13	13	13	13
	4	372	376	403	404	9	9	9	9
	5	375	377	409	413	12	12	11	11
4	6	438	442	474	477	11	11	11	11
	0	27	28	29	29	5	5	5	5
	1	126	126	133	129	7	7	7	7
	2	218	218	224	220	5	5	6	5
	3	343	344	354	351	6	6	6	5
	4	413	414	426	425	6	6	6	6
	5	472	473	495	497	7	7	7	6
	6	865	865	911	942	10	10	9	9

Table 4.2.c. New estimates of total food ration (g) of **saithe** by age group and quarter of the year (1991), and associated coefficients of variation (CV) obtained by boot-strapping. Gastric evacuation model was applied to stomach content data at different levels of aggregation (haul, ICES square, round fish area, and total North Sea).

		Estimate				CV			
		Sample	Square	Round	Total	Sample	Square	Round	Total
quarter	age								
1	3	213	216	225	225	11	11	10	10
	4	313	321	335	339	12	12	13	13
	5	530	536	553	650	17	17	20	18
	6	765	769	743	999	17	17	18	17
	7	957	960	889	1151	21	21	22	20
	8	1140	1142	1054	1205	25	25	24	22
	9	3711	3711	3139	3142	79	79	88	88
2	3	1018	1036	1088	1088	7	7	7	7
	4	1263	1284	1350	1350	7	7	7	7
	5	2276	2296	2402	2402	6	6	5	5
	6	3895	3930	4165	4165	9	9	8	8
	7	5453	5501	5821	5821	9	9	8	8
	8	7133	7183	7464	7464	9	9	7	7
	9	8905	8943	9238	9243	9	9	8	8
3	3	614	615	621	621	17	17	16	16
	4	959	963	957	957	13	13	14	14
	5	1172	1173	1198	1207	14	14	14	14
	6	1289	1282	1352	1380	20	20	19	20
	7	3115	3104	3161	3210	15	15	12	13
	8	3940	3931	3992	4042	15	15	13	14
	9	5229	5239	5353	5359	19	19	17	16
4	3	713	697	656	656	7	7	7	7
	4	1101	1096	1086	1086	10	10	10	10
	5	1808	1836	1981	1985	17	17	19	19
	6	2053	2075	2392	2404	22	21	20	19
	7	2426	2445	2801	2820	29	28	24	23
	8	3053	3089	3350	3373	35	35	29	28
	9	3755	3781	4069	4084	26	26	23	23

Table 4.2.d. New estimates of total food ration (g) of **whiting** by age group and quarter of the year (1991), and associated coefficients of variation (CV) obtained by boot-strapping. Gastric evacuation model was applied to stomach content data at different levels of aggregation (haul, ICES square, round fish area, and total North Sea).

		Estimate				CV			
		Sample	Square	Round	Total	Sample	Square	Round	Total
quarter	age								
1	1	21	21	23	23	7	7	6	6
	2	48	48	53	51	6	6	5	5
	3	65	66	71	69	7	7	6	6
	4	83	84	89	87	7	7	6	6
	5	87	88	93	91	7	7	6	6
	6	109	109	115	113	7	7	7	7
2	1	71	71	72	71	6	6	5	5
	2	135	135	134	134	3	3	3	3
	3	175	175	175	173	3	3	3	3
	4	208	208	208	207	3	3	3	3
	5	225	225	225	224	3	3	3	3
	6	229	229	229	228	3	3	3	3
3	0	23	23	24	26	6	6	6	5
	1	90	90	93	92	5	5	6	6
	2	199	198	188	178	3	3	3	3
	3	245	244	228	215	3	3	3	3
	4	269	268	251	237	3	3	3	3
	5	296	295	275	260	3	3	3	3
4	6	363	362	343	330	5	5	5	4
	0	38	38	39	40	3	3	3	3
	1	129	129	128	124	4	4	4	4
	2	180	180	175	171	5	5	5	5
	3	245	244	233	230	5	5	5	5
	4	243	242	231	228	5	5	5	5
	5	275	274	258	255	4	4	4	4
	6	267	266	252	248	4	4	4	5

Table 4.3. New estimates of total food ration (g) by age group and quarter of the year (1991) expressed relative to old estimates for each gadoid predator.

Predator	age	Quarter			
		1	2	3	4
Cod	0	.	.	1.68	1.33
	1	0.99	0.69	1.25	0.94
	2	1.26	1.39	1.85	1.52
	3	1.76	1.47	1.55	1.58
	4	2.16	1.78	1.63	1.45
	5	1.98	2.01	1.40	2.02
Haddock	6	1.73	2.13	1.85	2.34
	0	.	.	0.95	0.80
	1	1.03	1.15	1.03	1.24
	2	0.94	1.02	1.06	0.97
	3	0.92	0.92	1.13	1.09
	4	0.93	0.89	0.82	1.00
Saithe	5	0.80	0.65	0.65	0.92
	6	1.00	0.47	0.64	1.43
	3	0.50	1.08	0.41	0.60
	4	0.49	0.94	0.46	0.68
	5	0.61	1.28	0.42	0.85
	6	0.67	1.69	0.36	0.77
Whiting	7	0.67	1.91	0.72	0.75
	8	0.67	2.15	0.78	0.82
	9	1.91	2.35	0.92	0.90
	0	.	.	0.83	0.76
	1	0.48	0.55	0.47	0.68
	2	0.43	0.52	0.57	0.58
	3	0.38	0.49	0.51	0.58
	4	0.36	0.44	0.44	0.47
	5	0.32	0.40	0.41	0.46
	6	0.35	0.37	0.46	0.40

Table 4.4. New estimates of food ration by prey group (g) of gadoid predator by age group and quarter of the year (1991). Gastric evacuation model was applied to stomach content data at different levels of aggregation (haul, ICES square, round fish area, and total North Sea).

Predator: cod; quarter 1

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	5	6	6	7	22	22	21	20	26	27	27	25
Annelida	7	7	6	6	39	38	34	35	58	54	49	48
Anomura mm	3	4	4	4	34	34	32	29	56	55	48	45
Astacidea	0	0	0	0	6	6	6	5	15	15	12	11
Caridea	16	15	14	14	21	20	16	14	23	22	16	15
Cephalopoda	0	0	0	0	14	14	12	11	29	27	22	21
Clupea harengus	0	0	0	0	31	33	41	41	172	181	200	198
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	0	0	0	0	1	1	1	1	0	0	0	0
Gadus morhua	1	2	2	2	18	25	24	25
Limanda limanda	0	0	0	0	14	15	15	15	83	85	90	84
Melanogr. aeglefinus	11	10	10	10	58	55	52	51
Merlangius merlangus	0	0	0	0	63	65	76	85	317	328	382	391
Other	0	0	0	0	2	2	3	4	0	0	0	0
Other Crustaceans	1	1	0	0	2	2	1	1	2	2	1	1
Other Inv.	4	3	3	3	12	12	11	12	13	13	9	9
Other fish	15	15	16	16	137	137	146	145	400	395	425	416
Pleuronectes platessa	1	1	2	2
Solea solea	5	5	6	5	21	21	24	22
Sprattus sprattus	4	4	7	7	2	2	3	4	6	7	9	10
Trisopterus esmarki	1	1	1	1	68	69	86	92	170	180	190	188
Total	56	56	58	59	486	491	521	531	1470	1493	1584	1561

Predator: cod; quarter 1

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	69	75	89	75	119	134	179	154	147	168	237	204
Annelida	91	88	85	78	93	92	86	77	88	88	78	68
Anomura mm	96	94	72	63	114	112	81	70	115	113	78	68
Astacidea	34	35	25	26	50	53	40	46	61	65	49	59
Caridea	29	27	23	19	19	16	15	13	10	8	8	7
Cephalopoda	97	96	118	156	287	286	356	472	418	417	519	688

Predator: cod; quarter 1

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Clupea harengus	482	497	584	579	660	683	833	854	765	792	979	1024
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	0	0	0	0
Gadus morhua	66	85	91	82	79	86	106	74	94	96	125	78
Limanda limanda	339	348	355	300	622	648	623	500	789	827	775	605
Melanogr. aeglefinus	106	112	98	101	124	152	137	150	135	176	161	179
Merlangius merlangus	784	809	907	906	1110	1132	1260	1265	1322	1340	1487	1495
Other	0	0	0	0	1	1	0	0	1	1	0	0
Other Crustaceans	2	2	1	1	4	3	2	1	5	4	2	1
Other Inv.	12	12	7	6	10	10	9	7	9	9	10	7
Other fish	838	821	1047	1048	981	960	1219	1239	1051	1028	1277	1311
Pleuronectes platessa	8	9	10	9	13	14	15	13	15	16	16	14
Solea solea	50	50	52	47	64	66	63	57	64	67	59	54
Sprattus sprattus	14	14	16	15	13	13	13	12	10	10	9	8
Trisopterus esmarki	245	248	220	220	247	246	214	217	253	249	213	217
Total	3363	3424	3800	3732	4611	4707	5251	5222	5351	5473	6081	6088

Predator: cod; quarter 2

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	19	21	26	28	144	149	172	182	254	259	275	287
Annelida	12	12	10	9	59	57	52	48	109	106	106	94
Anomura mm	25	25	26	26	73	73	68	64	126	125	118	101
Astacidea	0	0	1	1	21	21	23	24	36	36	33	34
Caridea	5	5	4	4	10	9	7	6	12	12	7	7
Cephalopoda	2	2	2	2	3	2	2	2	5	4	4	3
Clupea harengus	0	0	0	1	75	81	101	109	390	412	519	556
Echinodermata	0	0	0	0	3	3	2	2	10	7	6	4
Euphausiacea	0	0	0	0	7	8	10	10	4	5	6	6
Gadus morhua	0	0	0	0	5	5	5	4	29	29	23	20
Limanda limanda	1	1	1	1	95	97	99	94	376	381	394	376
Melanogr. aeglefinus	0	0	0	0	27	28	25	25	108	108	106	115
Merlangius merlangus	0	0	0	0	62	62	66	68	203	202	214	220
Other	0	0	0	0	3	3	2	2	5	5	3	3
Other Crustaceans	15	15	13	12	20	18	14	14	7	6	5	5

Predator: cod; quarter 2

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Other Inv.	17	18	18	18	34	36	41	43	41	42	47	50
Other fish	24	24	22	22	224	224	207	206	606	607	569	574
Pleuronectes platessa	0	0	1	1
Solea solea	1	1	1	1	7	7	7	6	31	32	31	29
Sprattus sprattus	0	0	0	0	3	3	3	3	9	9	10	9
Trisopterus esmarki	6	7	8	9	201	205	227	222	327	327	360	368
Total	130	131	134	134	1076	1091	1133	1134	2688	2715	2837	2863

Predator: cod; quarter 2

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	453	482	526	599	597	649	730	833	597	634	691	780
Annelida	245	240	200	186	448	443	338	324	558	555	491	425
Anomura mm	248	245	234	197	360	354	336	302	321	319	287	253
Astacidea	164	171	158	151	376	394	370	354	680	699	665	653
Caridea	8	8	6	5	11	11	10	9	7	7	6	6
Cephalopoda	5	5	5	4	7	7	7	5	77	77	43	37
Clupea harengus	926	950	1129	1147	1421	1436	1597	1553	1826	1837	2117	2152
Echinodermata	11	9	4	3	5	4	2	2	3	3	2	1
Euphausiacea	2	2	3	3	3	3	3	3	2	2	2	2
Gadus morhua	194	195	229	266	387	389	550	688	252	254	371	469
Limanda limanda	754	757	801	796	1194	1194	1276	1301	1537	1542	1680	1780
Melanogr. aeglefinus	394	392	463	490	790	785	969	987	1165	1193	1417	1477
Merlangius merlangus	495	498	504	510	766	794	787	774	1504	1512	1701	1827
Other	8	8	8	8	15	15	16	18	15	15	15	15
Other Crustaceans	2	2	1	1	1	1	0	0	0	0	0	0
Other Inv.	96	104	108	123	218	245	257	297	149	168	177	205
Other fish	1140	1127	1074	1112	1380	1334	1262	1314	1592	1523	1412	1423
Pleuronectes platessa	43	45	61	70	125	133	180	206	87	92	125	142
Solea solea	96	96	100	98	126	126	138	139	516	536	622	577
Sprattus sprattus	11	11	11	10	22	22	20	18	32	32	31	34
Trisopterus esmarki	367	359	364	367	455	445	400	382	1070	1039	911	838
Total	5663	5708	5989	6147	8707	8785	9247	9509	11989	12040	12766	13099

Predator: cod; quarter 3

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	0	0	0	0	91	91	100	109	122	122	120	111
Annelida	1	1	1	1	30	30	30	28	96	90	75	65
Anomura mm	1	1	1	1	47	48	45	37	100	98	87	72
Astacidea	0	0	0	0	4	4	5	5	20	20	20	17
Caridea	8	8	8	6	7	6	6	6	7	6	5	6
Cephalopoda	1	1	1	2	4	3	3	3
Clupea harengus	2	2	3	4	62	67	102	144
Echinodermata	0	0	0	0	0	0	0	0	2	1	1	1
Euphausiacea	0	0	1	1	1	1	1	1	0	0	0	0
Gadus morhua	0	0	0	0	13	14	13	14	47	45	36	32
Limanda limanda	0	0	0	0	1	1	1	1	94	95	90	86
Melanogr. aeglefinus	0	0	0	0	11	11	11	12	101	103	95	99
Merlangius merlangus	20	20	22	20	74	77	72	69
Other Crustaceans	6	6	6	8	1	1	1	1	1	1	0	0
Other Inv.	0	0	0	0	10	10	11	8	15	15	16	11
Other fish	7	7	7	6	38	37	33	34	292	286	254	240
Pleuronectes platessa	0	0	0	0	0	0	0	0
Solea solea	1	1	1	1
Sprattus sprattus	2	2	3	2	6	8	11	10
Trisopterus esmarki	0	0	0	1	40	41	47	54	273	282	324	321
Total	23	23	24	23	317	319	331	337	1315	1320	1313	1287

Predator: cod; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	214	213	190	169	222	218	162	136	64	55	20	19
Annelida	168	159	89	79	199	198	93	80	106	106	84	84
Anomura mm	174	171	143	113	250	246	219	133	292	293	281	175
Astacidea	50	48	38	33	213	203	143	105	282	282	206	123
Caridea	11	11	7	6	35	35	23	16	42	42	24	23
Cephalopoda	9	4	5	6	36	12	18	21	0	0	0	0
Clupea harengus	276	302	486	577	1152	1238	1637	1939	1417	1458	1947	2288

Predator: cod; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Echinodermata	5	3	2	2	8	8	7	5	18	17	15	10
Euphausiacea	0	1	0	0	0	0	0	0	0	0	0	0
Gadus morhua	54	52	30	29	33	25	14	12	25	17	8	9
Limanda limanda	236	238	216	188	439	439	299	332	826	826	507	629
Melanogr. aeglefinus	168	173	136	141	44	46	37	41	52	56	44	51
Merlangius merlangus	217	234	216	197	433	460	409	270	620	650	578	414
Other Crustaceans	0	0	0	0	1	1	0	0	2	2	0	0
Other Inv.	15	15	15	11	3	3	3	2	0	0	0	0
Other fish	403	395	319	265	794	765	608	509	1176	1130	803	716
Pleuronectes platessa
Solea solea	1	1	1	1	0	0	0	0
Sprattus sprattus	47	48	71	56	20	21	24	14	6	6	9	7
Trisopterus esmarki	416	412	443	419	652	618	575	573	422	418	251	291
Total	2466	2480	2409	2292	4536	4536	4271	4187	5351	5358	4778	4840

Predator: cod; quarter 3

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	33	27	6	
Annelida	851	851	642	646
Anomura mm	682	683	635	510
Astacidea	180	180	131	78
Caridea	27	27	15	15
Cephalopoda				
Clupea harengus	1165	1190	1531	1858
Echinodermata	11	11	10	
Euphausiacea	1	1	1	
Gadus morhua	14	9	4	33
Limanda limanda	999	1000	788	1399
Melanogr. aeglefinus	633	635	638	379
Merlangius merlangus	386	404	359	253
Other Crustaceans	2	2	0	
Other Inv.	115	115	129	24
Other fish	2057	2027	1931	2815
Pleuronectes platessa				
Solea solea				

Predator: cod; quarter 3

	age			
	6			
	Sample	Square	Round	Total
Sprattus sprattus				
Trisopterus esmarki	1204	1202	1202	473
Total	8360	8364	8025	8497

Predator: cod; quarter 4

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	0	0	0	0	20	20	22	24	66	68	70	72
Annelida	15	15	16	16	26	23	18	16	108	106	69	64
Anomura mm	3	3	4	4	48	48	48	45	122	122	116	106
Astacidea	0	0	0	0	4	4	3	3	11	11	12	13
Caridea	29	30	30	31	52	52	47	43	81	81	76	64
Cephalopoda	0	0	0	0	2	2	1	1	25	25	28	19
Clupea harengus	1	1	2	2	36	41	56	56
Echinodermata	0	0	0	0	2	2	2	2	6	6	4	4
Euphausiacea	1	1	1	1	3	3	2	2	9	8	7	7
Gadus morhua	0	0	0	0	8	8	7	7	13	12	9	9
Limanda limanda	0	0	0	0	5	5	5	5	123	123	134	127
Melanogr. aeglefinus	0	0	0	0	67	70	76	82	248	251	246	256
Merlangius merlangus	0	0	0	0	37	37	42	47	238	243	267	348
Other	0	0	0	0	1	1	1	1	5	5	4	2
Other Crustaceans	1	1	1	1	1	1	1	1	2	3	3	2
Other Inv.	4	4	4	4	10	10	10	11	14	12	12	11
Other fish	15	15	16	15	83	83	74	70	338	335	331	312
Pleuronectes platessa
Solea solea	0	0	0	0	10	12	14	14
Sprattus sprattus	0	0	0	0	7	8	13	14	7	7	9	7
Trisopterus esmarki	0	0	0	0	89	93	103	108	349	353	367	362
Total	70	70	72	73	466	472	477	483	1812	1824	1835	1854

Predator: cod; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	138	138	143	140	289	289	316	326	388	389	442	485
Annelida	167	166	108	110	152	153	102	105	48	48	29	32
Anomura mm	178	180	168	153	221	224	209	187	174	176	166	127
Astacidea	24	24	26	26	30	29	26	27	28	27	17	19
Caridea	72	69	60	53	75	71	59	53	47	45	34	29
Cephalopoda	39	39	40	30	24	24	22	20	7	6	6	6
Clupea harengus	99	116	156	159	125	152	200	205	2038	2048	2809	3634
Echinodermata	15	15	11	11	15	15	11	11	5	5	4	4
Euphausiacea	5	5	4	4	3	2	1	1	3	2	1	2
Gadus morhua	16	15	14	12	16	17	18	12	23	23	26	16
Limanda limanda	539	535	566	509	1010	1005	1042	888	1684	1682	1672	1337
Melanogr. aeglefinus	495	489	478	485	698	683	660	685	1163	1153	981	861
Merlangius merlangus	484	498	538	639	869	893	991	1078	959	966	1127	1225
Other	18	16	15	6	32	29	25	7	13	12	10	2
Other Crustaceans	5	5	5	1	9	9	9	1	5	5	4	1
Other Inv.	39	35	36	37	75	69	74	80	82	79	89	101
Other fish	684	687	691	669	1019	1027	1050	1023	1995	2002	1969	1520
Pleuronectes platessa	863	863	950	847
Solea solea	20	22	25	21	25	25	27	19	16	16	17	12
Sprattus sprattus	12	12	13	8	19	19	23	13	46	46	48	40
Trisopterus esmarki	684	682	674	685	1035	1020	958	1033	1447	1438	1144	1147
Total	3733	3747	3769	3757	5741	5755	5824	5777	11031	11033	11546	11446

Predator: cod; quarter 4

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	287	289	331	370
Annelida	32	31	17	19
Anomura mm	150	151	142	96
Astacidea	21	21	12	14
Caridea	29	28	20	16
Cephalopoda	2	2	2	1
Clupea harengus	3699	3701	5085	6606
Echinodermata	5	5	4	4
Euphausiacea	2	2	1	1

Predator: cod; quarter 4

	age			
	6			
	Sample	Square	Round	Total
Gadus morhua	18	18	20	12
Limanda limanda	2073	2072	2032	1676
Melanogr. aeglefinus	1557	1552	1225	960
Merlangius merlangus	751	750	878	937
Other	3	3	2	0
Other Crustaceans	2	2	1	1
Other Inv.	56	55	64	73
Other fish	2500	2504	2442	1659
Pleuronectes platessa	1595	1595	1756	1566
Solea solea	7	7	8	6
Sprattus sprattus	71	72	72	66
Trisopterus esmarki	1507	1506	1072	960
Total	14366	14366	15187	15042

Predator: haddock; quarter 1

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	1	1	2	2	4	4	6	7	6	7	10	12
Annelida	13	13	14	14	31	31	28	28	51	50	49	47
Anomura mm	0	0	1	1	2	2	2	2	5	5	5	6
Astacidea	0	0	0	0	1	1	1	1	1	1	1	1
Caridea	2	2	2	2	3	3	3	3	5	5	5	5
Cephalopoda	2	2	4	4	5	5	7	7	3	3	3	3
Clupea harengus	0	0	0	0	0	0	1	1
Echinodermata	3	3	3	3	5	5	4	4	8	8	6	6
Euphausiacea	0	0	0	0	0	0	0	0	0	0	0	0
Other	0	1	1	1	1	1	1	1	1	1	1	1
Other Crustaceans	2	2	2	2	4	4	4	4	9	9	8	8
Other Inv.	4	4	3	3	9	9	9	9	9	9	8	7
Other fish	2	2	2	2	9	9	10	10	16	16	17	18
Solea solea	0	0	0	0	0	0	0	0	1	1	0	0
Trisopterus esmarki	0	0	0	0	1	1	1	1	8	9	14	15
Total	30	30	33	33	74	74	76	77	122	122	129	130

Predator: haddock; quarter 1

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	4	4	5	6	4	4	5	6	1	1	2	2
Annelida	65	63	62	58	69	68	64	61	84	83	75	73
Anomura mm	8	8	8	9	8	9	9	9	9	10	10	10
Astacidea	1	1	1	1	1	1	1	1	1	1	1	1
Caridea	6	6	6	6	6	6	6	6	7	7	6	6
Cephalopoda	4	4	4	4	4	4	4	4	4	4	4	4
Clupea harengus	1	1	4	7	1	1	4	8	2	2	6	11
Echinodermata	14	14	11	11	17	17	15	15	30	30	27	27
Euphausiacea	0	0	0	0	0	0	0	0	0	0	0	0
Other	2	2	1	1	2	1	1	1	2	2	1	2
Other Crustaceans	13	13	11	11	13	14	12	12	20	20	20	21
Other Inv.	9	9	8	8	9	9	8	8	16	16	15	15
Other fish	24	24	26	27	29	29	33	34	72	72	80	83
Solea solea	1	1	0	1	1	1	0	1	0	1	0	0
Trisopterus esmarki	20	20	29	31	23	24	32	35	35	35	46	50
Total	169	169	176	181	188	188	195	201	284	284	294	304

Predator: haddock; quarter 2

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	16	17	24	24	41	42	61	64	53	54	86	91
Annelida	10	10	7	7	25	25	16	16	38	38	25	24
Anomura mm	0	0	0	0	1	1	1	1	3	3	4	4
Astacidea	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	1	1	1	1	1	2	1	1	2	2	2	1
Cephalopoda	0	0	0	0	0	0	0	0
Clupea harengus
Echinodermata	3	3	2	2	12	12	8	8	27	27	19	18
Euphausiacea	1	1	1	1	2	2	3	3	3	3	4	4
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Maurolicus	0	0	0	0	0	0	0	0	1	1	1	1
Other	1	1	1	1	2	2	1	1	2	2	2	2
Other Crustaceans	46	47	49	50	83	84	88	87	108	109	119	119
Other Inv.	6	6	5	5	14	14	14	14	23	23	24	25
Other fish	9	9	7	7	13	13	10	10	16	16	15	15
Pleuronectes platessa	0	0	0	0	0	0	0	0

Predator: haddock; quarter 2

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Pollachius virens	0	0	0	0	0	0	0	0
Solea solea	0	0	0	0	0	0	0	0
Sprattus sprattus	0	0	0	0	0	0	0	0
Trisopterus esmarki	0	0	0	0	0	0	0	0	0	0	0	0
Total	96	96	98	99	196	197	205	206	277	280	302	305

Predator: haddock; quarter 2

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	68	70	108	116	62	64	103	110	49	50	77	83
Annelida	50	50	35	34	46	46	33	30	39	39	28	27
Anomura mm	6	6	8	7	5	5	7	7	4	4	6	6
Astacidea	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	3	3	3	2	3	3	3	2	3	3	2	2
Cephalopoda	0	0	0	0
Clupea harengus	0	0	0	0	2	2	3	3
Echinodermata	46	46	34	32	43	43	32	29	34	34	26	24
Euphausiacea	2	2	3	3	2	2	2	2	3	3	4	4
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Maurolicus	1	1	1	1	1	2	2	2	1	1	1	1
Other	5	5	4	4	4	4	3	3	3	4	3	3
Other Crustaceans	126	128	140	140	116	118	128	130	103	104	113	113
Other Inv.	30	30	33	36	28	28	30	32	23	23	25	27
Other fish	25	25	29	29	26	26	30	30	30	30	40	40
Pleuronectes platessa	0	0	0	0
Pollachius virens	0	0	0	0	0	0	0	0	0	0	0	0
Solea solea
Sprattus sprattus	0	0	0	1	0	0	0	0	0	0	0	0
Trisopterus esmarki	1	1	1	2	1	1	1	1	1	1	1	1
Total	363	368	402	408	337	341	374	380	295	297	330	334

Predator: haddock; quarter 3

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	1	1	1	2	46	47	69	80	82	84	120	140
Annelida	6	6	5	5	20	20	14	11	47	47	34	28
Anomura mm	0	0	0	0	1	1	1	1	4	4	4	3
Astacidea	0	0	0	0	0	0	0	0	0	0	1	1
Caridea	0	0	0	0	1	1	1	1	1	1	1	1
Cephalopoda	0	0	0	0	0	0	0	0	0	0	0	0
Echinodermata	0	0	0	0	5	5	4	3	24	24	20	17
Euphausiacea	1	1	1	1	0	0	0	0	1	1	0	0
Gadus morhua
Melanogr. aeglefinus	1	1	1	1	6	7	7	5
Other	0	0	0	0	1	1	1	1	3	3	2	2
Other Crustaceans	4	4	4	5	20	20	20	15	20	19	18	14
Other Inv.	1	1	1	2	7	7	5	5	24	24	21	21
Other fish	2	2	2	2	13	13	10	9	21	21	17	15
Solea solea	0	0	0	0	0	0	0	0
Trisopterus esmarki	0	0	0	0	1	1	2	2	8	9	15	12
Total	15	16	17	17	118	119	130	130	243	245	260	259

Predator: haddock; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	139	140	206	229	113	114	166	186	140	141	202	227
Annelida	75	76	57	52	74	75	55	50	69	69	52	46
Anomura mm	6	6	6	6	6	6	6	6	6	6	6	5
Astacidea	0	0	1	1	0	0	1	1	0	0	1	1
Caridea	1	1	1	1	1	1	1	1	2	2	1	1
Cephalopoda	0	0	0	0	0	0	0	0	0	0	0	0
Echinodermata	47	47	36	31	44	44	36	32	41	41	31	27
Euphausiacea	2	2	1	1	2	2	1	1	1	1	1	1
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Melanogr. aeglefinus	6	7	8	7	7	7	9	8	7	8	8	6
Other	5	5	4	3	5	5	4	3	4	4	3	3
Other Crustaceans	17	17	15	12	17	16	15	12	18	18	15	12
Other Inv.	48	49	46	44	58	60	58	56	36	36	33	31
Other fish	26	25	21	19	28	28	24	21	23	23	19	16
Solea solea

Predator: haddock; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Trisopterus esmarki	14	15	24	22	17	17	28	26	27	27	37	36
Total	386	390	424	427	372	376	403	404	374	377	409	412

Predator: haddock; quarter 3

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	140	141	198	220
Annelida	78	79	58	54
Anomura mm	7	7	7	7
Astacidea	0	0	0	0
Caridea	2	2	2	2
Cephalopoda	0	0	0	0
Echinodermata	49	49	40	35
Euphausiacea	2	2	1	1
Gadus morhua	0	0	0	0
Melanogr. aeglefinus	6	6	7	7
Other	5	5	4	3
Other Crustaceans	16	16	13	11
Other Inv.	56	58	55	53
Other fish	28	28	24	21
Solea solea
Trisopterus esmarki	47	47	62	62
Total	437	441	473	477

Predator: haddock; quarter 4

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	0	0	0	0	8	8	13	13	15	15	25	25
Annelida	11	11	11	10	36	36	28	26	73	73	56	53
Anomura mm	1	1	1	1	5	5	5	5	9	9	9	9
Astacidea	0	0	0	0	2	2	2	3	3	3	4	5
Caridea	1	1	2	2	3	3	3	3	3	3	2	2
Cephalopoda	0	0	0	0	1	1	2	2	1	1	2	1
Clupea harengus	0	0	0	0
Echinodermata	3	3	2	2	9	9	6	6	13	13	9	8
Euphausiacea	1	1	1	1	10	10	12	12	4	4	5	5
Melanogr. aeglefinus	0	0	0	0
Other	0	0	0	0	2	2	3	2	5	5	5	5
Other Crustaceans	5	5	6	6	8	8	7	7	13	13	11	11
Other Inv.	2	2	1	1	9	9	7	6	17	17	16	15
Other fish	3	3	4	4	14	14	14	13	18	18	18	17
Trisopterus esmarki	1	1	1	1	20	20	32	32	43	44	63	65
Total	27	28	29	29	126	126	133	129	218	218	224	220

Predator: haddock; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	18	18	28	27	20	20	29	29	20	20	28	29
Annelida	91	90	67	64	93	93	68	65	92	91	69	66
Anomura mm	16	16	15	15	19	19	18	18	22	22	22	22
Astacidea	3	3	4	4	3	3	4	4	2	2	2	2
Caridea	5	5	5	5	6	6	6	6	8	8	8	9
Cephalopoda	2	2	2	2	2	2	2	2	3	3	3	4
Clupea harengus	1	1	3	3	2	2	6	6	5	4	10	11
Echinodermata	20	19	15	14	25	25	20	17	29	29	24	21
Euphausiacea	3	3	3	3	4	4	3	4	5	5	5	5
Melanogr. aeglefinus	2	2	2	2	4	4	4	4	6	6	7	8
Other	18	18	18	18	25	25	24	24	36	35	34	33
Other Crustaceans	18	17	17	17	19	19	19	19	20	20	20	21
Other Inv.	38	38	38	38	44	44	45	44	54	54	55	55
Other fish	32	32	33	33	42	42	44	44	57	57	61	62

Predator: haddock; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Trisopterus esmarki	78	80	104	106	105	108	135	139	113	116	145	149
Total	343	344	354	351	413	414	426	425	472	473	495	497

Predator: haddock; quarter 4

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	41	41	49	51
Annelida	81	81	63	62
Anomura mm	33	33	31	33
Astacidea	1	1	1	1
Caridea	14	14	15	16
Cephalopoda	6	6	6	6
Clupea harengus	16	16	36	39
Echinodermata	74	74	55	50
Euphausiacea	4	5	3	3
Melanogr. aeglefinus	23	23	26	28
Other	70	70	59	57
Other Crustaceans	26	26	25	27
Other Inv.	44	44	42	43
Other fish	133	133	152	153
Trisopterus esmarki	298	299	345	373
Total	865	865	911	942

Predator: saithe; quarter 1

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	3	3	2	2	4	4	2	2	7	7	3	3
Annelida	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	0	0	0	0	2	2	1	1	9	9	4	4
Cephalopoda	1	1	0	0	3	3	1	1	3	3	1	1
Clupea harengus	1	1	1	1	23	23	40	40	112	113	197	197
Euphausiacea	19	23	25	25	31	39	45	46	13	14	13	25
Maurolucus	10	11	7	7	9	9	6	6	14	14	8	8
Other	0	0	0	0	0	0	0	0	0	0	0	0

Predator: saithe; quarter 1

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Other Crustaceans	25	23	12	12	30	27	13	13	18	17	7	6
Other Inv.	1	1	0	0	1	2	1	1	0	0	0	0
Other fish	64	62	56	56	92	89	79	80	88	85	65	94
Trisopterus esmarki	89	91	120	120	118	123	146	149	265	274	254	310
Total	213	216	225	225	313	321	335	339	530	536	553	650

Predator: saithe; quarter 1

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	4	4	2	2	1	1	1	1
Annelida	0	0	0	0
Caridea	10	10	7	7	9	9	7	7	5	5	4	4
Cephalopoda	2	2	1	1	1	1	0	0	0	0	0	0
Clupea harengus	99	99	166	166	130	131	201	201	218	218	314	314
Euphausiacea	6	6	4	36	4	3	2	35	2	1	1	20
Maurolicus	8	8	4	5	3	3	2	2
Other	0	0	0	0
Other Crustaceans	14	14	4	4	25	25	5	5	36	36	7	7
Other Inv.	0	0	0	0
Other fish	121	114	79	155	178	172	108	186	285	281	186	230
Trisopterus esmarki	501	512	476	624	605	615	564	715	593	598	542	630
Total	765	769	743	999	957	960	889	1151	1140	1141	1053	1204

Predator: saithe; quarter 1

	age			
	9			
	Sample	Square	Round	Total
Group				
Ammodytidae
Annelida
Caridea	1	1	0	0
Cephalopoda
Clupea harengus	2458	2458	2103	2103
Euphausiacea
Maurolicus
Other

Predator: saithe; quarter 1

	age			
	9			
	Sample	Square	Round	Total
Other Crustaceans	14	14	3	3
Other Inv.
Other fish	997	997	808	808
Trisopterus esmarki	211	211	195	199
Total	3680	3680	3109	3113

Predator: saithe; quarter 2

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	23	24	26	26	45	46	55	55	193	193	255	255
Caridea	0	0	0	0	0	0	0	0	0	0	0	0
Clupea harengus	0	0	0	0	2	2	3	3	100	102	107	107
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	542	561	606	606	632	653	695	695	850	868	861	861
Maurollicus	183	191	215	215	201	208	235	235	241	238	260	260
Melanogr. aeglefinus	0	0	0	0	2	2	2	2	7	8	7	7
Merlangius merlangus	0	0	0	0
Other Crustaceans	107	103	80	80	99	94	69	69	59	54	38	38
Other fish	73	67	69	69	103	96	98	98	247	246	225	225
Trisopterus esmarki	90	90	91	91	180	183	194	194	580	586	650	650
Total	1018	1036	1088	1088	1263	1284	1350	1350	2276	2296	2402	2402

Predator: saithe; quarter 2

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	187	187	251	251	180	179	180	180	382	382	312	312
Caridea
Clupea harengus	422	432	476	476	1096	1132	1289	1289	2240	2324	2675	2675
Echinodermata	0	0	0	0
Euphausiacea	1405	1440	1517	1517	1611	1646	1737	1737	1357	1366	1325	1325
Maurollicus	348	338	382	382	332	321	382	382	153	148	180	180
Melanogr. aeglefinus	3	3	3	3	135	135	157	157	439	439	509	509
Merlangius merlangus	1	1	2	2	120	116	114	114	385	372	365	365

Predator: saithe; quarter 2

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Other Crustaceans	35	34	15	15	30	30	8	8	23	23	8	8
Other fish	801	816	776	776	1271	1288	1254	1254	1540	1536	1495	1495
Trisopterus esmarki	693	678	744	744	679	654	702	702	613	593	594	594
Total	3895	3930	4165	4165	5453	5501	5821	5821	7133	7183	7464	7464

Predator: saithe; quarter 2

	age			
	9			
	Sample	Square	Round	Total
Group				
Ammodytidae	427	427	349	349
Caridea
Clupea harengus	4133	4239	4822	4822
Echinodermata
Euphausiacea	959	945	857	857
Maurolicus	4	3	2	2
Melanogr. aeglefinus	492	492	569	569
Merlangius merlangus	429	415	408	408
Other Crustaceans	14	15	8	8
Other fish	1880	1842	1695	1700
Trisopterus esmarki	567	566	527	527
Total	8905	8943	9238	9243

Predator: saithe; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	32	33	42	42	43	46	43	43	26	29	22	22
Caridea	1	1	1	1	8	8	8	8
Clupea harengus	1	1	1	1	24	24	34	34	72	76	98	98
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	66	68	73	73	36	37	37	37	45	45	41	41
Gadus morhua	0	0	0	0	12	12	18	18	11	11	16	16
Melanogr. aeglefinus	174	174	190	190	109	109	112	112	51	51	54	54
Merlangius merlangus	34	34	35	35	36	36	33	33	43	43	40	40
Other Crustaceans	7	7	5	5	30	30	17	17	25	25	14	14

Predator: saithe; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Other fish	79	78	61	61	171	168	119	119	171	169	136	137
Sprattus sprattus	1	1	1	1	1	1	0	0
Trisopterus esmarki	216	216	211	211	495	498	543	543	720	716	770	778
Total	612	613	619	619	959	963	957	957	1172	1173	1198	1207

Predator: saithe; quarter 3

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae
Caridea	15	15	15	15	4	4	4	4	0	0	0	0
Clupea harengus	149	162	202	202	1537	1567	2015	2015	2183	2214	2853	2853
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	64	64	59	59	53	53	26	26	54	55	16	16
Gadus morhua
Melanogr. aeglefinus	52	52	64	64	74	72	59	59	85	82	59	59
Merlangius merlangus	48	48	49	49	18	18	17	17	7	8	6	6
Other Crustaceans
Other fish	112	112	118	120	354	355	219	223	473	474	278	281
Sprattus sprattus
Trisopterus esmarki	849	828	845	870	1071	1031	816	862	1130	1091	774	819
Total	1289	1282	1352	1379	3111	3100	3156	3205	3934	3925	3985	4035

Predator: saithe; quarter 3

	age			
	9			
	Sample	Square	Round	Total
Group				
Ammodytidae
Caridea
Clupea harengus	3283	3291	4255	4255
Echinodermata	1	1	0	0
Euphausiacea	88	88	24	24
Gadus morhua
Melanogr. aeglefinus	109	104	71	71
Merlangius merlangus	4	4	3	3

Predator: saithe; quarter 3

	age			
	9			
	Sample	Square	Round	Total
Other Crustaceans
Other fish	706	707	423	424
Sprattus sprattus
Trisopterus esmarki	1040	1046	576	582
Total	5229	5239	5353	5359

Predator: saithe; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	0	0	0	0	0	0	0	0	0	0	0	0
Annelida	0	0	0	0	1	0	0	0	0	0	0	0
Caridea	2	3	3	3	2	2	2	2	4	3	3	3
Cephalopoda	0	0	0	0	0	0	0	0	0	0	0	0
Clupea harengus	4	4	4	4	66	63	73	73	463	455	523	523
Echinodermata	0	0	0	0	1	0	0	0	1	1	1	1
Euphausiacea	24	19	13	13	21	16	12	12	17	13	12	12
Maurolicus	0	0	0	0
Melanogr. aeglefinus	52	51	45	45	98	93	70	70	129	120	90	90
Merlangius merlangus	7	7	5	5	15	14	9	9	11	10	9	9
Other Crustaceans	1	1	0	0	1	1	0	0	0	0	0	0
Other Inv.	0	0	0	0	0	0	0	0
Other fish	104	95	77	77	142	133	118	118	235	222	233	234
Trisopterus esmarki	517	516	508	508	755	774	800	800	946	1011	1108	1111
Total	713	697	656	656	1101	1096	1085	1086	1806	1834	1979	1983

Predator: saithe; quarter 4

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae
Annelida
Caridea	2	2	2	2
Cephalopoda	0	0	0	0
Clupea harengus	917	918	1131	1131	1700	1719	2000	2000	2673	2709	2930	2930

Predator: saithe; quarter 4

	age											
	6				7				8			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Echinodermata	1	1	1	1	0	0	1	1	0	0	0	0
Euphausiacea	7	6	6	6	1	1	1	1	2	2	3	3
Maurolicus
Melanogr. aeglefinus	277	273	170	170	290	290	179	179	96	96	81	81
Merlangius merlangus	23	23	29	29	27	27	34	34	12	12	12	12
Other Crustaceans
Other Inv.
Other fish	206	198	265	266	126	126	200	205	74	74	95	104
Trisopterus esmarki	615	650	782	792	272	272	375	388	181	181	211	224
Total	2048	2070	2386	2398	2417	2436	2790	2809	3038	3074	3332	3354

Predator: saithe; quarter 4

	age			
	9			
	Sample	Square	Round	Total
Group				
Ammodytidae
Annelida
Caridea	1	1	1	1
Cephalopoda
Clupea harengus	2816	2841	3315	3315
Echinodermata
Euphausiacea	1	2	2	2
Maurolicus
Melanogr. aeglefinus	97	97	123	123
Merlangius merlangus	6	6	6	6
Other Crustaceans
Other Inv.
Other fish	47	47	55	61
Trisopterus esmarki	743	743	519	528
Total	3711	3737	4022	4037

Predator: whiting; quarter 1

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	4	4	5	6	5	5	7	6	5	5	6	5
Annelida	4	4	3	3	4	4	2	2	3	3	2	2
Anomura mm	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	2	2	2	2	3	3	3	2	2	2	2	2
Cephalopoda	1	1	1	1	3	3	3	3	5	5	6	5
Clupea harengus	0	0	0	0	1	1	1	1	2	2	2	2
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	1	1	1	1	0	0	0	0	0	0	0	0
Melanogr. aeglefinus	0	0	0	0	0	0	0	0	0	0	0	0
Merlangius merlangus	0	0	0	0	2	2	2	2	3	3	4	3
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	2	2	2	2	2	2	1	1	3	3	2	1
Other Inv.	2	2	2	2	3	3	2	2	3	3	3	2
Other fish	3	3	3	3	8	8	7	7	11	11	10	8
Pollachius virens	0	0	0	0	0	0	0	0
Sprattus sprattus	1	1	1	1	3	3	4	7	6	6	7	12
Trisopterus esmarki	1	1	2	2	14	14	19	18	21	21	27	25
Total	21	21	23	23	48	48	53	51	65	66	71	69

Predator: whiting; quarter 1

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	6	6	6	5	6	6	6	5	7	7	6	5
Annelida	3	3	2	1	3	3	2	1	2	2	1	1
Anomura mm	1	1	1	1	1	1	1	1	2	2	2	2
Caridea	2	2	2	2	2	2	2	2	3	3	2	2
Cephalopoda	5	5	6	5	5	5	6	5	4	4	5	4
Clupea harengus	3	3	3	3	3	3	4	3	4	4	6	5
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	0	0	0	0	0	0	0	0	0	0	0	0
Melanogr. aeglefinus	0	0	1	0	0	0	1	1	1	1	1	1
Merlangius merlangus	5	5	6	5	5	5	6	5	7	7	8	7
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	3	3	2	1	3	3	2	1	4	4	2	2

Predator: whiting; quarter 1

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Other Inv.	3	3	3	2	3	3	3	2	3	3	2	2
Other fish	17	16	14	12	18	17	15	13	25	24	22	19
Pollachius virens	0	0	0	0	0	0	0	0	1	1	1	1
Sprattus sprattus	6	7	8	12	7	7	8	12	7	7	8	10
Trisopterus esmarki	29	29	36	36	30	31	38	38	40	41	49	53
Total	83	84	89	87	87	88	93	91	109	109	115	113

Predator: whiting; quarter 2

	age											
	1				2				3			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	20	21	24	25	63	65	70	71	88	90	98	99
Annelida	15	15	14	13	6	5	4	4	5	4	3	3
Anomura mm	1	1	1	1	1	1	1	1	2	2	1	1
Caridea	1	1	1	1	1	1	1	1	1	1	1	1
Cephalopoda	1	1	2	2	1	1	1	1	1	1	1	1
Clupea harengus	0	0	0	0	1	1	1	1	2	2	2	2
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	0	0	0	0	1	2	2	1	3	3	3	3
Gadus morhua	0	0	1	0	0	0	0	0	0	0	0	0
Limanda limanda	0	0	0	0	0	0	0	0
Maurolicus	0	0	0	0	0	0	0	0	1	1	1	1
Melanogr. aeglefinus	0	0	0	0	0	0	0	0	0	0	0	0
Merlangius merlangus	0	0	0	0	1	1	1	1	2	2	2	2
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	13	12	10	10	20	19	16	14	21	20	16	15
Other Inv.	9	9	8	8	10	10	7	7	10	9	7	7
Other fish	7	7	7	7	19	18	18	18	25	25	22	22
Sprattus sprattus	2	2	3	3	6	6	9	9	7	7	9	10
Trisopterus esmarki	0	0	0	0	3	4	4	3	7	8	8	7
Total	71	71	72	71	135	135	134	134	175	175	175	173

Predator: whiting; quarter 2

	age											
	4				5				6			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	104	107	116	117	110	112	122	123	107	109	118	120
Annelida	4	4	3	2	4	4	3	2	4	4	3	3
Anomura mm	2	2	1	1	2	2	2	2	2	2	2	2
Caridea	1	1	1	1	1	1	1	1	1	1	1	1
Cephalopoda	1	1	1	1	1	1	1	1	1	1	1	1
Clupea harengus	3	4	4	4	4	5	5	5	4	5	5	5
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	5	5	5	5	7	7	7	7	8	9	9	8
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Limanda limanda
Maurolicus	1	1	1	1	1	1	1	1	1	1	1	1
Melanogr. aeglefinus	0	0	0	0	0	0	0	0	0	0	0	0
Merlangius merlangus	3	3	3	3	4	4	3	3	5	5	4	4
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	21	20	16	14	21	20	15	14	20	19	15	13
Other Inv.	10	9	7	7	9	9	7	7	9	9	7	7
Other fish	30	29	25	26	32	31	27	27	31	30	26	27
Sprattus sprattus	8	9	11	12	9	10	12	13	10	11	13	14
Trisopterus esmarki	13	14	14	13	18	18	18	17	23	23	23	23
Total	208	208	208	207	225	225	225	224	229	229	229	228

Predator: whiting; quarter 3

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	3	3	4	6	18	18	21	26	45	45	44	43
Annelida	0	0	0	0	7	7	5	4	7	6	4	3
Anomura mm	0	0	0	0	8	8	8	7	8	8	6	5
Astacidea	0	0	0	0	0	0	0	0
Caridea	1	1	1	0	4	4	2	2	4	3	2	2
Cephalopoda	0	0	0	0	0	0	0	0	0	0	0	0
Clupea harengus	0	0	0	0	1	1	1	1	9	10	11	12
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	0	0	0	0	3	3	3	3	3	3	2	2
Gadus morhua	0	0	0	0	0	0	0	0

Predator: whiting; quarter 3

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Limanda limanda	0	0	0	0	0	0	0	0
Melanogr. aeglefinus	0	0	0	0	1	1	1	1	4	4	4	4
Merlangius merlangus	0	0	0	0	3	3	2	2	10	10	9	8
Other	0	0	0	0	0	0	0	0
Other Crustaceans	6	6	7	6	9	9	6	5	6	6	4	3
Other Inv.	1	1	1	1	8	8	7	6	9	9	7	6
Other fish	8	8	8	8	16	16	15	13	38	37	27	24
Sprattus sprattus	0	0	0	0	7	9	17	16	25	25	38	37
Trisopterus esmarki	3	3	3	4	4	4	4	5	30	30	30	28
Total	23	23	24	26	90	90	93	92	199	198	188	178

Predator: whiting; quarter 3

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	54	55	51	47	59	59	55	50	65	65	59	53
Annelida	8	8	3	3	9	9	4	3	10	10	4	3
Anomura mm	7	7	5	4	7	7	4	4	6	6	4	3
Astacidea	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	4	3	2	1	3	3	1	1	3	3	1	1
Cephalopoda	0	0	0	0	0	0	0	0	0	0	0	0
Clupea harengus	15	15	17	20	19	19	21	25	22	22	25	30
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	3	3	2	2	3	3	2	2	3	3	2	2
Gadus morhua	1	1	1	0	1	1	1	0	1	1	1	1
Limanda limanda	0	0	0	0	0	0	0	0
Melanogr. aeglefinus	6	6	6	5	8	8	7	6	9	9	8	7
Merlangius merlangus	12	12	11	9	13	13	12	10	14	14	13	11
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	5	5	3	2	5	4	2	2	4	3	2	2
Other Inv.	10	10	6	6	10	10	6	5	10	10	6	5
Other fish	44	43	30	27	46	44	31	28	49	48	34	29
Sprattus sprattus	28	29	41	41	28	29	40	40	30	31	42	42
Trisopterus esmarki	47	47	49	46	59	59	62	60	68	68	73	70
Total	245	244	228	215	269	268	251	237	296	295	275	260

Predator: whiting; quarter 3

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	77	77	70	65
Annelida	12	12	4	3
Anomura mm	5	5	3	2
Astacidea	0	0	0	0
Caridea	3	3	1	1
Cephalopoda	0	0	0	0
Clupea harengus	32	33	38	43
Echinodermata	0	0	0	0
Euphausiacea	4	4	2	2
Gadus morhua	1	1	1	1
Limanda limanda
Melanogr. aeglefinus	12	12	10	9
Merlangius merlangus	18	18	20	16
Other	0	0	0	0
Other Crustaceans	3	3	2	1
Other Inv.	9	9	5	5
Other fish	56	54	38	32
Sprattus sprattus	29	30	39	39
Trisopterus esmarki	101	101	109	111
Total	363	362	343	330

Predator: whiting; quarter 4

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	3	3	4	4	24	24	28	26	22	23	28	27
Annelida	1	1	1	1	4	4	3	2	4	5	3	3
Anomura mm	1	1	1	1	4	4	4	3	3	3	3	3
Astacidea	0	0	0	0	0	0	0	0
Caridea	4	4	4	3	7	7	6	5	7	7	5	4
Cephalopoda	3	3	3	3	1	1	1	1	1	1	1	1
Clupea harengus	0	0	0	0	2	2	2	2	2	2	2	3
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	3	3	3	3	13	13	10	6	14	14	11	5
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Limanda limanda	0	0	0	0	0	0	0	0	0	0	0	0
Maurolicus	0	0	0	0	0	0	0	0	0	0	0	0

Predator: whiting; quarter 4

	age											
	0				1				2			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Melanogr. aeglefinus	0	0	0	1	1	1	1	1	4	4	4	4
Merlangius merlangus	0	0	0	0	3	3	3	2	5	5	5	5
Other	0	0	0	0	0	0	1	0	0	0	0	0
Other Crustaceans	6	6	5	4	5	5	4	3	4	4	3	2
Other Inv.	1	1	0	0	2	2	2	2	3	3	3	2
Other fish	9	9	9	9	29	28	28	25	40	39	38	32
Solea solea	0	0	0	0	0	0	0	0	0	0	0	0
Sprattus sprattus	1	1	1	2	6	6	8	8	8	9	11	11
Trisopterus esmarki	6	6	7	10	30	30	30	38	62	62	58	68
Total	38	38	39	40	129	129	128	124	180	180	175	171

Predator: whiting; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Group												
Ammodytidae	21	21	26	27	21	21	26	27	20	20	25	25
Annelida	4	5	3	3	4	5	3	3	4	4	2	2
Anomura mm	3	3	2	2	3	3	3	2	3	3	2	2
Astacidea	0	0	0	0	0	0	0	0	0	0	0	0
Caridea	6	6	4	4	6	6	4	4	6	6	4	3
Cephalopoda	1	1	1	1	1	1	1	1	1	1	1	1
Clupea harengus	3	3	3	3	3	3	3	3	3	3	3	3
Echinodermata	0	0	0	0	0	0	0	0	0	0	0	0
Euphausiacea	13	13	10	4	13	13	10	4	11	11	8	4
Gadus morhua	0	0	0	0	0	0	0	0	0	0	0	0
Limanda limanda	0	0	0	0	0	0	0	0	0	0	0	0
Maurolicus	0	0	0	0
Melanogr. aeglefinus	10	10	10	10	10	10	10	10	16	16	15	15
Merlangius merlangus	9	9	9	9	8	8	9	8	10	10	10	9
Other	0	0	0	0	0	0	0	0	0	0	0	0
Other Crustaceans	3	3	2	2	3	3	2	2	3	3	2	1
Other Inv.	5	5	4	3	5	5	5	4	10	9	9	7
Other fish	51	50	47	40	50	49	47	39	53	51	47	40
Solea solea	1	1	0	0	1	1	0	0	1	1	0	0
Sprattus sprattus	10	11	14	13	10	11	14	13	10	10	13	12

Predator: whiting; quarter 4

	age											
	3				4				5			
	Sample	Square	Round	Total	Sample	Square	Round	Total	Sample	Square	Round	Total
Trisopterus esmarki	105	105	97	109	103	103	95	108	125	125	116	128
Total	245	244	233	230	243	242	231	228	275	274	258	255

Predator: whiting; quarter 4

	age			
	6			
	Sample	Square	Round	Total
Group				
Ammodytidae	21	21	26	26
Annelida	4	4	2	2
Anomura mm	3	3	2	2
Astacidea	0	0	0	0
Caridea	6	6	4	3
Cephalopoda	1	1	1	1
Clupea harengus	3	3	3	3
Echinodermata	0	0	0	0
Euphausiacea	11	11	8	4
Gadus morhua	0	0	0	0
Limanda limanda	0	0	0	0
Maurolicus
Melanogr. aeglefinus	13	13	13	14
Merlangius merlangus	11	11	11	10
Other	0	0	0	0
Other Crustaceans	3	3	2	2
Other Inv.	6	6	5	4
Other fish	53	51	47	40
Solea solea	1	1	0	0
Sprattus sprattus	10	11	13	12
Trisopterus esmarki	122	122	112	124
Total	267	266	252	248

5 CURRENT KNOWLEDGE OF THE HABITAT REQUIREMENTS OF COMMERCIAL, RARE AND THREATENED FISH SPECIES

5.1 Introduction

Assessing the habitat requirements for any given fish species is reliant on descriptive or quantitative information on the geographic range of all life-history stages at an appropriate spatial and temporal resolution (Schmitt, 1999), including:

- Current and historical population (or stock) size and the bio-geographical distribution
- Spatio-temporal patterns in the distribution and relative abundance of the various life-history stages and their biological and habitat requirements
- Ecological and environmental characteristics of the habitats, and their spatial distribution
- Population density, survivorship, growth rate, breeding success and production rates for the various habitats used

Data that are required to assist in the interpretation of the relative importance of habitats include long-term data sets of habitat use, an improved knowledge of the distribution and ecological stability of aquatic habitats, and improved knowledge of fish behaviour in specific habitats (Able, 1999). Important habitat requirements for fish include spawning, nursery, pupping and feeding grounds and migratory corridors.

The quality of available data varies greatly among species, habitats and geographical areas. Studies that would assist in the interpretation of the potential role of “Essential Fish Habitat” include post-larval studies, understanding of spawning behaviour, the role of offshore reefs (geological and biological) and other hard substrates as spawning grounds and refuges, the effect of complex habitats on survivorship of various life-history stages in the field, site fidelity, and the locations of high abundance of threatened species. It may also be important to be able to delineate and map important habitats for management purposes.

In the western North Atlantic, the North East Fisheries Science Centre (NEFSC) of the National Marine Fisheries Service (NMFS) has been proactive in describing the habitats used by a variety of commercial fish and shellfish (Reid *et al.*, 1999a; <http://www.nefsc.noaa.gov/nefsc/habitat/efh/>). Although focusing on the western North Atlantic, many of the species discussed also occur in the eastern North Atlantic, and the general principles of habitat requirements may be comparable to European seas. This series of reports has included cod *Gadus morhua* (Fahay *et al.*, 1999a), haddock *Melanogrammus aeglefinus* (Cargnelli *et al.*, 1999a), saithe *Pollachius virens* (Cargnelli *et al.*, 1999c), halibut *Hippoglossus hippoglossus* (Cargnelli *et al.*, 1999b), witch *Glyptocephalus cynoglossus* (Cargnelli *et al.*, 1999d), long rough dab *Hippoglossoides platessoides* (Johnson *et al.*, 1999a), spurdog *Squalus acanthias* (McMillan and Morse, 1999), herring *Clupea harengus* (Reid *et al.*, 1999b) and mackerel *Scomber scombrus* (Studholme *et al.*, 1999).

Other western North Atlantic species for which habitats have been described, and for which there are sister taxa that support commercial fisheries in the eastern North Atlantic include various species of flatfish (Chang *et al.*, 1999a; Johnson *et al.*, 1999b; Packer *et al.*, 1999; Pereira *et al.*, 1999), hake (Chang *et al.*, 1999b,c; Morse *et al.*, 1999; Steimle *et al.*, 1999a), skate (Packer *et al.*, 2003a-g), redfish (Pikanowski *et al.*, 1999) and anglerfish (Steimle *et al.*, 1999b).

For many species of marine fish, their mean size increases with depth (Heincke’s Law). The ecological benefits of juvenile fish inhabiting shallower environments include that larger piscivorous fishes may be less abundant in shallow water and that nearshore environments may be structurally complex (Gotceitas *et al.*, 1997; Linehan *et al.*, 2001), with macrophytes and macroalgae providing refuges from predation.

Below a summary is given of some of the available information on important fish habitats. For many species the data available to the group were based on studies in the North Sea and adjacent waters. More integrated accounts using data from other ICES areas is required.

5.2 Cod-like fishes (Gadiformes)

Cod: Gadus morhua

Several inshore studies have shown the local importance of eelgrass and kelp habitats for cod *Gadus morhua* in sites off Newfoundland (e.g., Grant and Brown, 1998; Linehan *et al.*, 2001; Cote *et al.*, 2001), and there is evidence of density-dependent habitat selection (Grant and Brown, 1998). Laboratory studies have shown that juvenile cod undergo a shift in habitat use when predators are present (Gotceitas *et al.*, 1995; Fraser *et al.*, 1996), when they favour cobble habitats

(hiding in the interstitial spaces) or kelp (when cobble is unavailable). Field studies have confirmed that wild cod in Newfoundland waters exhibit habitat preferences, with juveniles preferring coarse substrates with high bathymetric relief (Gregory and Anderson, 1997), since such complex habitats providing suitable refuges. Hence, the behavioural strategies exhibited by cod in such coastal zones (limited inshore movements, shoaling during the day and preference for complex habitats) may play an important role in predator avoidance (Grant and Brown, 1998). Although settlement of cod may not differ between habitat types, the survivorship of post larvae and juveniles is greater in complex habitats (Tupper and Boutilier, 1995). Indeed, experimental field studies have shown that predation on juvenile cod is higher in deeper water and, within shallow water, lower in vegetated areas (Linehan *et al.*, 2001).

The above information is based on available information for cod in the western North Atlantic, and may differ substantially from cod stocks in other areas, such as the North Sea. In the North Sea, juvenile cod have in the 1980's and early 1990's been observed in high abundances, especially in winter, in rather unstructured habitats, i.e., shallow areas with low salinity waters in the Thames estuary (Riley and Parnell, 1983) and German Bight (Heessen and Daan, 1994). Apart from a low salinity, these areas are also characterised by relatively high temperatures in summer and autumn, and low temperatures in winter. A low abundance of predators and a high abundance of prey organisms in these areas may play an important role.

Hake (Merluccidae)

There is a lack of biological and ecological information on juvenile hake *Merluccius merluccius* over much of its range, although some information is available. Hake has important nursery grounds in localized areas off the Iberian shelf, with the processes of hake recruitment leading to well defined patches of juveniles in the Iberian shelf. Concentrations vary in density according to year-class strength, although they remain relatively stable in spatial location. In the Cantabrian Sea Shelf, years of massive inflow of the eastward shelf edge current produce low recruitment, due to larvae and eggs being transported away to the open sea (ICES, 2004). Several authors have studied recruitment processes and the reproductive biology of hake on the Portuguese continental shelf, supporting the ICES Assessment Working Group (ICES, 2004). Two important and well defined hake nursery areas are closed to trawling during the recruitment and spawning times (Cardador, *et al.*, 1997).

Related species in the western North Atlantic have been subject to detailed studies. In the Western Atlantic silver hake *Merluccius bilinearis* has important nursery grounds on the outer continental shelf at depths of 60-95m (Steves *et al.*, 2000; Steves and Cowen, 2000), although the highest densities of 0-group silver hake occur on substrates with a high proportion of amphipod tube cover (Auster *et al.*, 1997). Other topographic features such as sand waves can also form important habitats for this species (Auster *et al.*, 2003) and red hake *Urophycis chuss* are known to use shelly substrates for shelter (Steimle *et al.*, 1999a).

5.3 Flatfish (*Pleuronectiformes*)

Most flatfishes are associated with finer sediments, rather than with hard substrata. The structure of the sea bed is an important factor controlling their distribution and there is increasing evidence that flatfish species can distinguish between and select sediments on the basis of their grain size (Gibson, 1994).

The dependence on sediment is probably related both to the distribution of suitable prey and to the ability of flatfish to bury themselves. Burying provides some protection from predators and reduces the use of energy. The juvenile habitat is often a small and generally shallower part of the total habitat occupied by the species. For certain species nursery areas play an important role, whereas for other species no specific nursery areas are known. In general for North Atlantic flatfish the magnitude of recruitment is mainly an effect of transport to and quality of areas for larval development (van der Veer *et al.*, 1990, 2000, Beverton and Iles 1992; Bailey 1994; Wennhage and Pihl 2001).

The advantages for juveniles of living in shallower water could be higher temperatures, lower predation risk, abundant food and appropriate substrata. In most species there is a positive relation between length and depth, but there are species where juveniles seem to occur in deeper parts than the adults, such as witch *Glyptocephalus cynoglossus* (Powells and Kohler 1970; Burnett *et al.* 1992).

Many flatfish species show a gradual offshore movement of juveniles as they grow. This might indicate that habitat quality for flatfish is size-dependent. Another common pattern is the annual micro- and macroscale movements and migrations between spawning, feeding, and wintering areas (e.g., Molander 1925; Aro 1989; Gibson 1997). A brief summary of the habitat requirements of the dominant commercial flatfish in the ICES area is given below, with the information available mostly for the North Sea and adjacent area. Information on juvenile habitats is also available for Norwegian waters (for further details see Albert *et al.* 1998).

Plaice: Pleuronectes platessa

Plaice spawn in distinct areas offshore and show spawning area fidelity (Gibson, 1997; Hunter *et al.*, 2003). In the North Sea, plaice spawning grounds are recognized over most of the shallower parts of the southern North Sea and off the east coast of Britain from Flamborough Head to the Moray Firth. Centres of high egg production are the eastern English Channel and the Southern Bight, while egg production around the Dogger Bank and in the German Bight is more diffuse (Harding *et al.*, 1978; Heessen and Rijnsdorp, 1989). Eggs and larvae drift towards shallow coastal areas. The Kattegat plaice stock is dependent on transport from the Skagerrak (Nielsen *et al.* 1998) and it has been suggested that in some years the recruits of plaice in the Skagerrak and the major parts in the Kattegat originate from a parental plaice stock in the North Sea (Ulmestrand 1992). In the Baltic plaice spawn only in the relatively saline water of the western Baltic and the deeper areas in the middle Baltic (Nissling *et al.* 2002, Mielck and Künne 1932, Molander 1964).

Duration of the planktonic egg and larval stages of plaice (three to four months) is long compared with that of, for instance, sole (about one month). Eggs and larvae are therefore for a long time exposed to residual currents, and the young plaice may settle in areas far away from the spawning area. Settling on the bottom happens only after complete metamorphosis, at a medium length of 13–14 mm (Russell, 1976; Modin *et al.* 1996).

Plaice make selective use of tidal currents in various stages of their life (e.g., de veen 1978). Metamorphosing larvae enter estuarine nursery areas on the flood tide (Creutzberg *et al.*, 1977); juvenile plaice in the Wadden Sea move with the flood tide onto sandy flats to feed and move back to the surrounding channels on the ebb tide (Kuipers, 1973).

Coastal and inshore stretches along the North Sea coasts of Britain and the continent are essential as nursery areas, but the Wadden Sea is considered to be the most important one (Kuipers, 1977). Also the shallow coastal areas in Skagerrack and Kattegat are important nursery habitats (Pihl *et al.* 2000). One-year-old plaice show a strictly coastal distribution but the older age groups gradually disperse further offshore, away from the nursery areas (Rijnsdorp, 1989). Recently, migration to deeper water seems to start at an earlier stage than approximately 10 years ago (Grift, pers. comm.).

Dab: Limanda limanda

Spawning occurs throughout the south-eastern North Sea, with offshore concentrations of eggs in the German Bight, north of the Frisian Islands, along the southern edge of the Dogger Bank (Land 1991; Rijnsdorp *et al.* 1992) and Northeast of Flamborough Head (Harding and Nichols 1987).

Settlement of postlarvae occurs at a size of 13–20 mm, shortly before metamorphosis is completed (Russell 1976), in shallow open bays along the coasts of Scotland and Wales (Macer 1967; Edwards and Steele 1968; Gibson 1973; Poxton *et al.* 1983). In the North Sea settlement occurs both in coastal and offshore waters (Bolle *et al.* 1994). They do not enter the intertidal region.

Although 0-group dab seem to be mainly concentrated in coastal areas below depths of 10 m (Bolle *et al.* 1994), small specimens have been caught at offshore stations in the Dogger Bank area and in the eastern central North Sea in February (Rijnsdorp *et al.* 1992). They gradually move into deeper waters as they grow, and those measuring over 11 cm are to be found mainly in waters deeper than 20 m (Poulsen, 1933). Nurseries seem to be very widespread.

Dab has an extended spawning period (4 months compared to 2 months in plaice) Feb-May in Kattegat and March-June in the Belt of Skerries (SD22) (Molander, 1964, Lagenfelt and Svedäng, 1999). Dab has partly adapted to the lower salinity in the Baltic and successful spawning may occur in the Sound (SD 23), the Arcona basin (SD 24) and occasionally in the Bornholm basin (SD 25) (Nissling *et al.* 2002).

Long rough dab: Hippoglossoides platessoides

Eggs or larvae have been found along the southern and northern border of its range in the North Sea (Economou, 1993; van der Land, 1991) and spawning is expected to occur in any area where long rough dab live (Russell, 1976).

Flounder: Platichthys flesus

Flounder occur in less saline waters and closer to the shore in shallower water than other flatfishes (Lagenfelt and Svedäng, 1999; Voigt, 2002, Molander, 1964). Flounder migrate between coast and sea, spawn in deeper waters and feed in shallow waters (Ehrenbaum, 1909, Molander, 1964, Aro, 1989). A common pattern in mainland northern Europe (Kerstan, 1991; Robin, 1991; Moeller and Dieckwisch, 1991) and at occasions reported from the British Isles

(Hutchinson and Hawkins, 1993) is that metamorphosing post-larvae flounder migrate to river estuaries and even far into main rivers during spring (see also Jager, 2001). The juveniles are found at higher densities in the low salinity areas and smaller fish prefers less saline water. Although the eggs are normally pelagic the flounder has in the northern Baltic evolved a reproductive strategy where the eggs develop at the bottom while the larvae are pelagic during early summer. This means that in the Baltic flounder spawns both on the deeper areas (with pelagic eggs) and in shallow areas (with demersal eggs) (Sandman, 1906, Solemdal, 1971, Mielck and Künne, 1932, Mielck, 1926, Nissling *et al* 2002).

Lemon sole: Microstomus kitt

The centre of distribution of lemon sole in the North Sea is situated in Scottish, Orkney, and Shetland waters. Mature lemon sole occur in small numbers throughout most of the remainder of the area, but not in south-eastern coastal waters. Smaller lemon sole occur in relatively high numbers only in the Moray Firth. Juveniles seem to prefer coarse stony substrates. Their catch rates with the main survey gears are so low, however, that we have a rather poor picture of the preferred habitats. It is assumed that the lemon sole spawns wherever it is found, but spawning is heaviest in places where it is caught in large numbers (Rae, 1965).

Greenland halibut: Reinhardtius hippoglossoides

This is a deep-water flatfish seldom caught shallower than 300m depth. In the North Sea area its distribution is confined to the slope of the Norwegian Sea and the northernmost part of the Norwegian Deep. This occurrence is part of a continuous distribution of adults and large juveniles from Faeroe Island and Iceland to Barents Sea and Svalbard. A nursery area is found in the Arctic, eastwards from Svalbard.

Atlantic halibut: Hippoglossus hippoglossus

The Atlantic halibut is distributed in both the northwest and northeast Atlantic, including the North Sea and the Skagerrak/Kattegat area. It is a deep sea fish, seldom caught shallower than 300m. Spawning takes place either at deep areas close to shore, including fjords, or on offshore sea banks of 300–1000 m depth. Juveniles occur on shallower coastal areas from 50m and downwards (Curry-Lindahl, 1985; Carlstrand 2002).

Turbot: Psetta maxima

The main spawning grounds for turbot in the North Sea are the Aberdeen Bank (rectangle 43E8) and Turbot Bank (rectangle 43E9) in the north (Rae and Devlin, 1972), while major spawning concentrations in the south occur around and to the north of the Dogger Bank. Several other smaller spawning grounds are recognized throughout the North Sea (Jones, 1970). In the Baltic turbot has adapted to the lower salinity and spawns successfully up to the Åland Sea both on the shallow banks and in the coastal areas (Molander 1964; Curry-Lindahl 1985; Andersson 1998; Muus *et al.* 1999; Voigt 2002).

Turbot spawn in shallow waters (10–40m in Skagerrak/Kattegat, 10–15 m in central Baltic) and the larvae migrate close to shore to very shallow water (down to one meter depth) (Curry-Lindahl, 1985; Voigt, 2002; Molander, 1964; Iglesias *et al.*, 2003).

Turbot eggs in the North Sea hatch at a mean size of 2.8 mm and larvae metamorphose at a mean size of 23 mm (Jones, 1972). The young are dispersed to suitable nursery areas during their pelagic phase and appear in shallow, knee-deep waters between June and October. One- and two-group juveniles of up to 30 cm are mostly found in waters less than 14 m in depth, while fish of 30–45 cm move into waters of 21–50 m depth. Larger specimens are confined to even deeper waters (Rae and Devlin, 1972; Jones, 1973). In the Baltic turbot is generally smaller (females mature at about 20 cm) but follow the same depth distribution. Young fishes prefer shallower water and sand/stone bottoms while older fishes are found in deeper waters in sandy or sand/clay mixed substrates (Molander, 1964).

Brill: Scophthalmus rhombus

Brill is a species of which our knowledge of its biology and habitat preferences is very limited. In the southern North Sea newly hatched young appear in the very shallow waters along the sandy beaches of the Dutch coast in July and August. One- and two-year-old brill share this shallow strip of sea with the 0-group specimens, but most brill have moved to deeper waters by the age of three (Lucio, 1986). Brill larvae hatch at 3.8mm and metamorphose at 17 mm (Jones, 1972). Tagging studies in Kattegat have shown that brill only perform short migrations to deeper water during autumn and winter, and return to the same shallow water in spring (Bagge 1987).

Megrim: Lepidorhombus whiffiagonis

In the North Sea megrim is seldom caught in waters less than 100 m deep, and catches only become substantial in the deepest parts of the North Sea. No separate nursery or spawning grounds are recognized in the North Sea (Rae, 1970).

Sole: Solea solea

Sole is generally considered to be a nocturnal feeder, spending the day buried in fine sediments (Kruuk, 1963; Lagardère, 1987). The sole is a southern species, and its distribution in the North Sea is confined to areas with relatively high bottom temperatures. Seasonal movements are probably temperature-induced because they are directed offshore into deeper, warmer waters during winter and inshore in March/May. Sole occur regularly in the Skagerrak and Kattegat, and occasionally in the western parts of the Baltic Sea (Molander, 1964, Muus *et al.*, 1999)

Sole may suffer high mortalities during extremely cold winters (Woodhead, 1964). Under such circumstances dense concentrations are formed in deeper, warmer waters like the Silver Pit (rectangles 35F0 and 36F0).

Peak spawning occurs in April/May, in coastal areas within the 30 m depth contour. Major centres of spawning in the southern North Sea are the Belgian coast, the Thames estuary, the Norfolk Banks, the area off Texel, and the German Bight (Rijnsdorp *et al.*, 1992; Russell, 1976). The pelagic larvae develop into demersal sole in a relatively short time, approximately one month. Consequently, passive transport of the larvae by the currents occurs over short distances only and it is likely that local abundance of 0-group sole mirrors the spawning success of local spawning groups (Beek *et al.*, 1989).

North Sea nurseries are situated in shallow waters all along the continental and English coasts at 5 to 10 m depth. Their relative contribution to the total North Sea sole recruitment may vary from year to year, but the nurseries in estuarine areas are always of minor importance (Beek *et al.*, 1989). As in plaice there is a positive relation between fish size and depth.

5.4 Essential Fish Habitat in the Baltic Sea

One of the objects of the Coastal and Marine Baltic Sea Protected Areas designated by HELCOM is to protect areas that are important nursery and spawning areas for fish. Many of the biotopes on HELCOMs Red List of marine and coastal biotopes and biotopes complexes of the Baltic Sea, Belt Sea and Kattegat are important for fishes (HELCOM, 1998).

In the coastal zone of the Baltic Sea, the brackish waters allow fresh-water fishes to be common, except in the southern parts where the salinity is high enough to restrict their distribution to estuaries and fresh-water lagoons. Perch (*Perca fluviatilis*), pike (*Esox lucius*) and pikeperch (*Stizostedion lucioperca*) are all abundant, widespread, and important for both commercial and recreational fisheries. Spawning and nursery areas are, however, limited because of generally low temperatures restricting recruitment for most populations to the shallowest and most sheltered parts. The most important habitats can be found within archipelagos, inlets, bays and freshwaters (Karås, 1999; Sandström, 2003).

An EU interregional project (Interreg IIIa) is studying the production of juvenile fish in shallow sea bays in the northern Baltic Proper and several projects launched by the Swedish National Board of Fisheries aim to map the important spawning and recruitment areas along the Swedish Baltic Sea Coast.

5.5 Case study: Preliminary identification of juvenile habitats of offshore demersal fishes (CEFAS)

Although the use of inshore nursery grounds by many species of fish (e.g., plaice, sole) is well documented, there are fewer studies examining the distribution and ecology of juvenile fishes that have nursery grounds offshore. The preliminary identification of some of the areas favoured by offshore species was made by the examination of CEFAS groundfish surveys (overall time period: 1977–2003) that operate in the eastern English Channel, Irish Sea, North Sea, Bristol Channel and Celtic Sea. The species considered were anglerfish *Lophius piscatorius*, hake *Merluccius merluccius*, lemon sole *Microstomus kitt* and megrim *Lepidorhombus whiffiagonis*. The data illustrated are for positive catches only, and are intended simply to highlight the general geographical areas where juveniles of these species are known to occur. More detailed field studies of these regions may facilitate more accurate descriptions of the types of habitat that are used as nursery grounds for these species.

The predominant depth range for juvenile hake (≤ 10 cm) was 80–160 m, and sites with a high abundance of small hake included off south-western Brittany, Celtic Sea and a small area west of the Isle of Man (Figure 5.1), although the large catches in the latter area were almost exclusively based on tows undertaken in 1988.

Juvenile lemon sole (≤ 10 cm) generally occurred in waters 20–90 m deep, and important areas were north of the Shetland Islands, north Norfolk coast, Dover Straits, northern Bristol Channel (off the Gower Peninsula and Pembrokeshire), and offshore sites in the Irish Sea and St George's Channel (Figure 5.1). All these sites comprise coarse grounds.

Juvenile anglerfish (≤ 15 cm) were relatively widespread (Figure 5.1), albeit at very low abundance, and the most consistent catches were made north of the Shetland Islands (130–220 m depth), Bristol Channel (30–80 m) and edge of the Celtic Sea (150–200 m depth).

Juvenile megrim (≤ 10 cm) were occasionally recorded on the edge of the continental shelf north of the Shetland Islands, although they were primarily taken along the edge of the continental shelf in the Celtic Sea, at depths of 100–220 m (Figure 5.1).

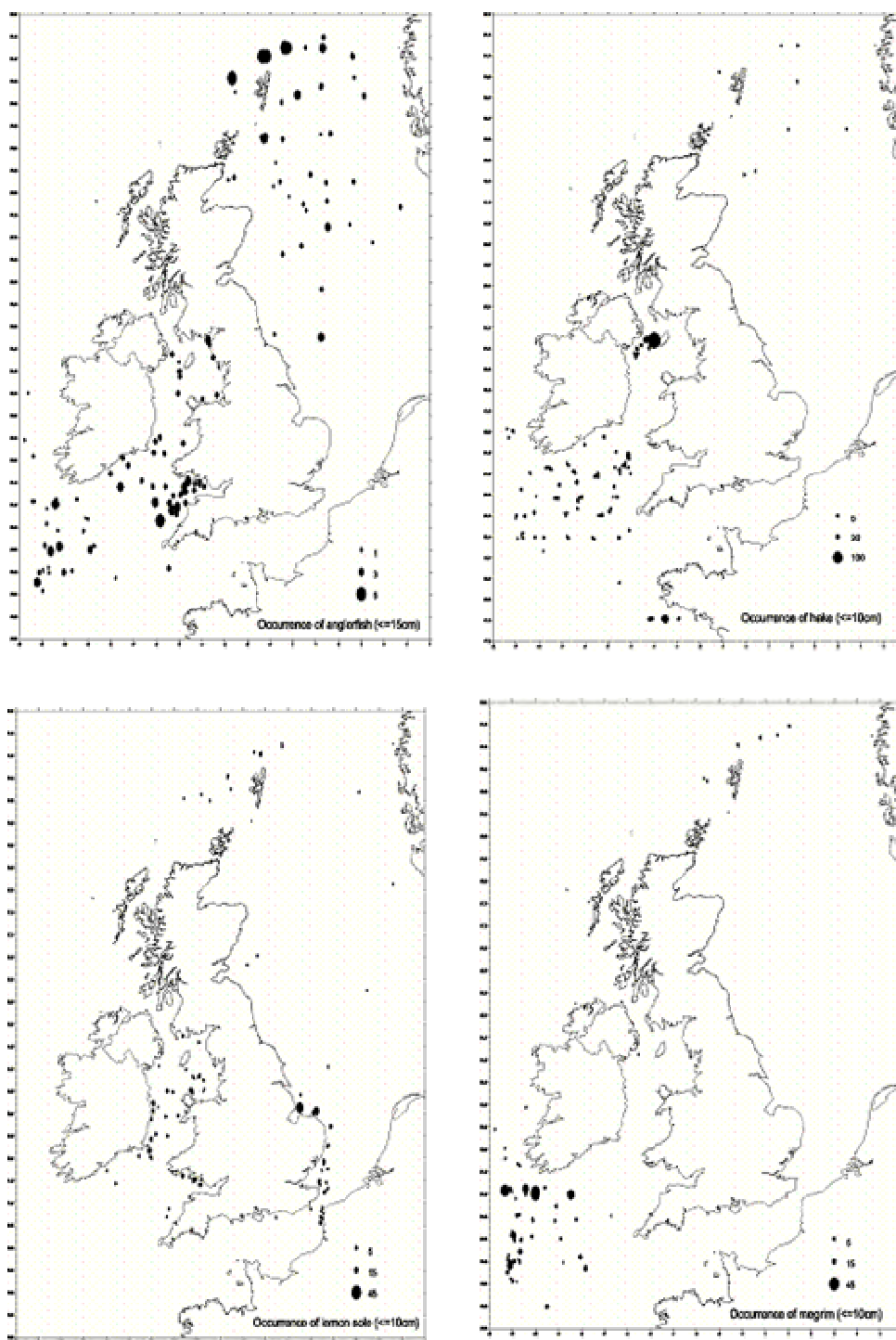


Figure 5.1. Occurrence of juvenile anglerfish, hake, lemon sole and megrim from CEFAS groundfish surveys.

5.6 Conclusions

Further research is needed to increase our understanding of the role of habitat in distinct parts of the life history of fishes, especially in spawning and the juvenile stage. Some of the major aspects in this should be:

- Existing survey information should be analysed to better describe juvenile distribution. For demersal species GIS databases could be used to link distribution with substrate and bathymetry.
- For shallow coastal areas, several national and international surveys (e.g., the Demersal Fish Survey along the English east coast and along the continental coasts of the south-eastern North Sea) are carried out on an annual basis that could be used to describe the importance of these waters.
- Field studies could examine the regional importance of complex habitats factors affecting survivorship and growth of juvenile fish.
- Spawning grounds for different species should be more adequately described from the results of egg and larval surveys (e.g., the PLACES project to identify spawning areas of cod and plaice in the North Sea). Data from commercial fisheries could provide further information on the distribution of spawning fish.

5.7 References

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6 RELATIVE CATCHABILITIES OF COMMON FISH SPECIES IN DIFFERENT SURVEY GEARS

6.1 Introduction

Different gears/survey methods provide different qualitative and/or quantitative estimates of the relative abundance of different fish species. Within European waters, the primary method of assessing the distribution and relative abundance of marine fishes is by trawl survey. Although there has been much progress in standardising gears in internationally-coordinated surveys, there is no single gear that will sample all components of the fish community with the same effectiveness. Hence, several gear types are used across Europe, and all these gears have different catchabilities for the various target and non-target fish.

Some of the major factors that will affect the size and species composition of trawl catches include mesh size, trawl opening (headline height, wing spread) and gear geometry, towing speed, vessel, number of warps and type of ground gear (e.g., Gordon and Bergstad, 1992; Wantiez, 1996; Vázquez, 2002). In addition to the differences that may be related to the type of survey gear, and its rigging, the efficiency of any trawl varies between species (Wantiez, 1996). It is also important to note that there are many factors that influence the catchability of a fish species to any single gear (see Ehrich, 1988, 1991; Ehrich and Gröger, 1989; Buijse *et al.*, 1992; Michalsen, *et al.*, 1996; Aglen *et al.*, 1997; Godø *et al.*, 1999; Swain *et al.*, 2000; Winger *et al.*, 2000; Ehrich and Stransky 2001; Adlerstein and Ehrich, 2002). These factors include the nature of the fishing protocol (e.g., tow duration and trawl speed) and fishing ground (e.g., sediment and seabed topography) and behaviour of fish species, which may vary in relation to biotic factors such as population density and size, and environmental conditions such as habitat type, temperature, light, turbidity and tidal conditions. Within the Baltic Sea, surveys of coastal fish populations use gillnets, and the sampling design, and type of gillnet and mesh sizes have major impacts on the number of fish species caught and their size spectrum (Appelberg *et al.*, 2003).

Several studies have compared towed gears with visual census (e.g., submersible or camera sledge), with slow-moving, conspicuous, benthic species often better sampled by visual census, and species that are either faster moving, more cryptic, or less-associated with the seabed itself better sampled by trawl (e.g., Uzmann *et al.*, 1977; Eleftheriou and Basford, 1984). Studies on rockfish in the Gulf of Alaska have found different population densities between visual census and trawl for various species. Estimated densities for those species that inhabited flat, pebble habitats were

greater in trawl surveys, due to the herding effect of the trawl, whereas trawl surveys underestimated the density of those species with a clear preference for boulder habitats (Krieger, 1993; Krieger and Sigler, 1996; Krieger and Ito, 1999; Else *et al.*, 2002). Although visual census may have advantages in certain habitats (e.g., rocky grounds that cannot be trawled efficiently), trawl surveys are the most efficient method of examining broadscale patterns on fine to coarse grounds.

Due to the factors discussed above, it is apparent that survey gear, and survey design, will have profound effects on our perception of the fish community. Differences in the species caught, the species composition and the size distribution of each species will have implications for studies on the diversity, structure and function of fish assemblages. Obviously, for describing the overall diversity of a site, the use of multiple gears is important. Neudecker and Breckling (1992) examined the differences in catches between stow net and 3m beam trawl in the Wadden Sea, with these two gear types capturing 30 and 19 species respectively. The type(s) of gear used will also effect the determination of biological parameters, such as size at age (e.g., Lucena and O'Brien, 2001).

Many laboratories have access to historical data sets that, in theory, could be used in analyses of contemporary versus historical data, with which to examine long-term changes. These analyses are often limited, however, as gears and vessels have changed over time, and the implications of different gears are a significant factor (e.g., Fock *et al.*, 2002).

Ehrich (1987a) examined differences between the GOV trawl and the '180 foot herring trawl', and reported conversion factors for the catches of haddock and cod. Other studies that have compared different gears have focused on the major differences in the species composition in the catches. For example, Merrett *et al.* (1991) examined differences in the catch rates of deep-water fishes between German bottom trawl, Granton Trawl and semi-balloon otter trawl. Although the species present were similar overall, there were major differences in catch rates. For example, arrowtooth eels *Synaphobranchus* spp., which accounted for only 2.5% of the catch of the German bottom trawl and Granton Trawl, were the dominant species in the catches of the semi-balloon otter trawl (46% of the catch). Conversely, squaloid sharks, smooth-heads (Alepocephalidae), *Hoplostethus* spp. and black scabbardfish *Aphanopus carbo* were better sampled by German bottom trawl and Granton Trawl.

Another important issue for fisheries scientists is intercalibrating between similar gears where there are differences in rigging. Engås *et al.* (1988) examined the catches of a Norwegian bottom sampling trawl using (a) rockhopper ground gear and (b) bobbins ground gear. They reported that the rockhopper ground gear was more efficient for catching cod and haddock, and this may be due to either the rockhopper ground gear forming a sand cloud and/or having better ground contact. Similarly, Ehrich (1987b) noted that juvenile cod can escape under gears with a bobbin ground gear, and that the capture efficiency for juvenile cod was highly dependent on the type of ground gear.

6.2 Case studies examining the relative catchabilities of gears

There is an increasing trend towards standardised surveys, where strictly-defined gears are used in internationally-coordinated surveys. In such circumstances, the general premise is that the gear specification is strictly adhered to. Nevertheless, methods of estimating the relative catchabilities of different gears are required for various reasons.

- It may enable better integration of data collected from different gears
- There may be the potential to extend time-series by comparing the gears used in contemporary surveys with those that were used in earlier surveys
- It is sometimes considered appropriate to change aspects of gears (e.g., the ground gear) to be able to fish different grounds

Two working documents were presented, and the results from these studies are included below.

6.2.1 Case study 1: Vulnerability of bottom fish species to the standard GOV

This text is adapted from the working document submitted by Ehrich *et al.*

6.2.1.1 Introduction

Traditionally, catch data from the IBTS have been used in stock assessments calculating indices of the recruiting as well as exploited year-classes of the eight target species in the North Sea. In light of the growing overfishing problems and the integration of the environment into the fishery policy during the last decade, the demands on surveys have generally increased. Now long-term data series like the IBTS are necessary to answer questions related to changes in bottom fish

assemblages due to fishing or other human activities and to climate change. Appropriate time series are scarce, as the gear or parts of the gear that affect the catch efficiency have changed during the period. This paper provides preliminary data that illustrates the degree to which qualitative and quantitative descriptors of fish assemblages in an area depend on the gear used, and provides information on the vulnerability (or relative catchability) of bottom fish species between different gears.

The difference between the terms catchability and vulnerability is explained by King (1995), who stated that “the distinction between vulnerability, the proportion of fish in the gear’s area of influence which is retained, and catchability, the proportion of fish in the stock which is caught by one unit of effort”. Vulnerability (or relative catchability) can vary between 0 (no specimen of a certain species which stays within the path of the gear is retained) and 1 (all specimens are caught).

The areas in which the comparison fishing trials took place are two of the 12 standard areas of the German Small-scale Bottom Trawl survey (GSBTS) which are distributed over the whole North Sea. When ever possible the German research vessels visit these areas and especially Box A, which is situated in the inner German Bight (Figure 6.2.1.1). In December 2001 and January 2002 comparative fishing trials were organized between the research cutter “Solea”, using a 7m-beam trawl and an otter trawl (Cod trawl) and the W. Herwig III, using a GOV and a 2m-beam trawl. In August 2003 there was an opportunity that both vessels stayed for 3 and 2 days in that area, overlapping by one day. In January 2004 the W. Herwig III visited Box A and Box N to monitor seasonal changes in the bottom fish and epibenthic assemblages. Additionally to the standard monitoring programme, the vulnerability of epibenthic and small bottom fish species was investigated by using a triple 2m-beam trawl instead of a single 2m-beam trawl as usual (Figure 6.2.1.2). This experiment was carried out within the scope of the EU-project “MAFCONS” (“Managing Fisheries to Conserve Groundfish and Benthic Invertebrate Species Diversity”).

6.2.1.2 Material and Methods

First experiment:

Within 7 days in December 2001, 54 hauls were conducted aboard “Solea”; 27 by each net, an otter-trawl (Standard Cod Trawl) and a 7m-beam trawl. The position of hauls and the towing directions were randomly distributed within Box A, an area of 10 to 10 nm in the German Bight around 25nm north-west of Helgoland. Each day the gear was changed to avoid the effect of changing environmental conditions during the period. The towing time and speed for these gears were 30 min and 3.5 knots respectively.

Two weeks later (4.–7.1.2002) the FRV “Walther Herwig III” also visited Box A to investigate the species distribution of fish and epibenthos using the GOV standard gear (19 hauls) and 2m-beam trawl (9 hauls). The GOV trawl was towed for 30 min at 4 knots, and the 2m-beam trawl for 5 min at one knot (Table 6.2.1.1).

During the period from mid December to the beginning of January the winter was mild and the hydrographic conditions changed very little (bottom temperature in December and in January between 6.5 and 7°C). Only small fishing activities were observed during that time of the year between Christmas and New Year. Therefore one can expect that the abundance of the bottom species (not pelagic species) were comparable.

Second experiment:

In August 2003 during the third quarter IBTS and the International Beam Trawl Survey (BTS) there was a chance to compare the GOV and the 7m-beam trawl simultaneously by Walther Herwig III and Solea. The two vessels fished in Box A for 3 days and 2 days respectively with a one day overlapping (Table 6.2.1.1). Also the stations and towing directions were randomly distributed over the area of 100nm² like in experiment 1.

Third experiment:

In January 2004 during the standard GSBTS into the German Bight 6 hauls were carried out with a triple 2 m-beam trawl in addition to the standard single trawl to monitor the epibenthos. Three standard 2 m-beam trawls were tied one after the other by steel ropes of 6 m length (Figure 6.2.1.2). On the head line of the first one a net sonde was fixed to determine the exact point in time when the gear touched and left the bottom.

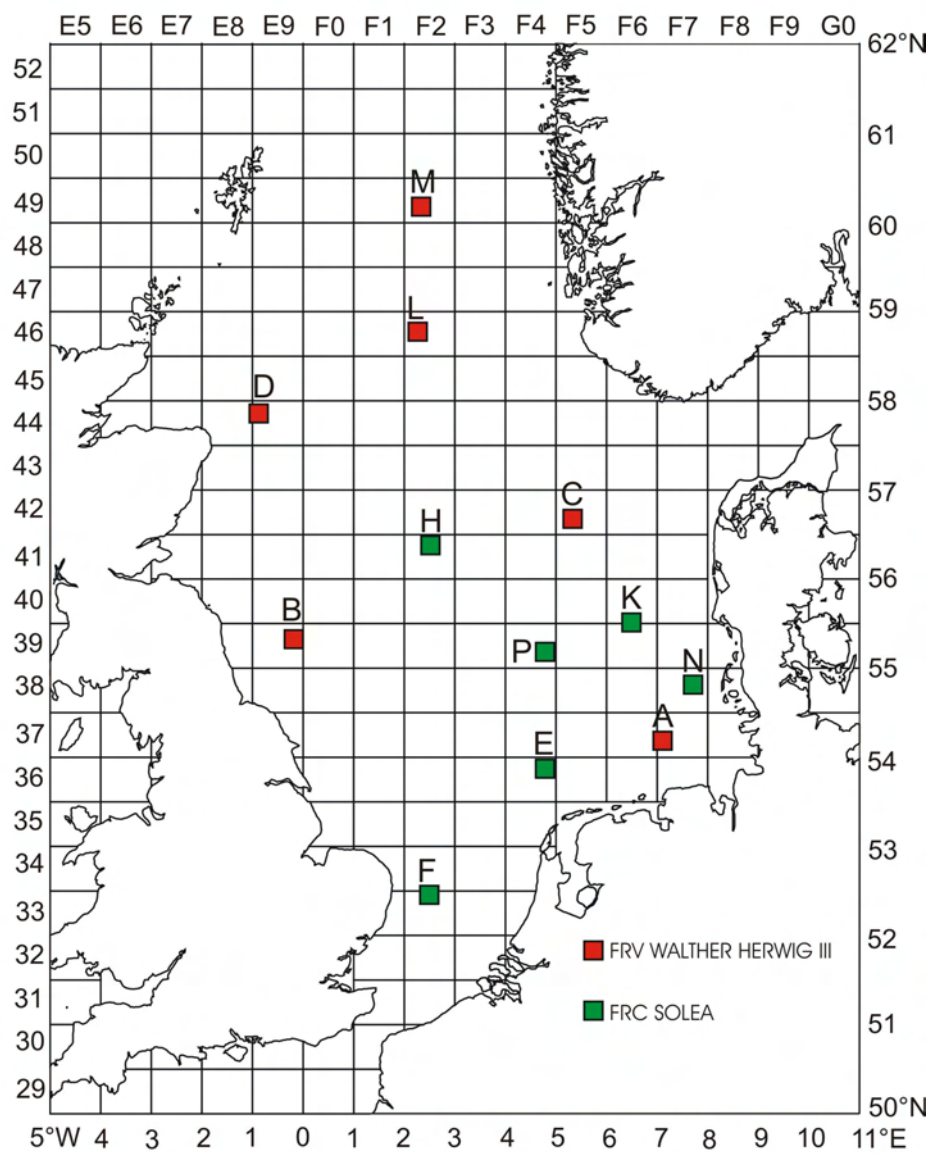


Figure 6.2.1.1. Position of the Boxes of the GSBTS.

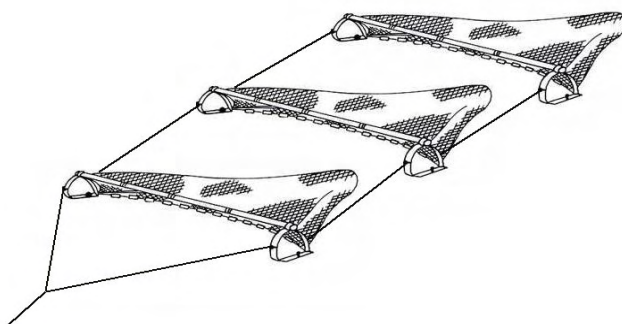


Figure 6.2.1.2. Triple 2m-beam trawl.

Gear description

GOV: The standard GOV equipped with the standard ground rope is exactly and fully described in the manual of the IBTS (ICES, 1999). Therefore only a short description is presented: At the given depths of around 40m in Box A the vertical and horizontal openings of the net were 4.5 m and 19 m respectively. The standard footrope comprised 20 cm rubber discs in the square and 10 cm rubber discs in the wings. For better bottom contact iron discs are fixed between the rubber discs to give extra weight. The codend is equipped with a fine mesh liner (20 mm).

Cod Trawl: The Standard Cod Trawl, used in the commercial fisheries in the early seventies and as a standard trawl aboard "Solea", has a headline length of 48.68 m and a circumference of 528 meshes at the bosom. At a depth of 40m the opening height of the net is 3.5 m and the horizontal wing spread 23 m. The net is provided with a rubber disc roller gear whose discs have a diameter of 20cm. The trawl is fitted out with 22 kg wing tip weights. The codend was as used in the GOV (Dahm *et al.*, 1996).

7m-beam trawl: The larger beam trawl is used in the International Beam Trawl Survey (IBS). It is characterised by a 7 m beam, 5 tickler chains and an overall net length of ca 21m. The normal 80 mm codend is fitted with a 20mm mesh liner.

2 m-beam trawl: The 2 m-beam trawl was developed within the scope of an EU-project to monitor the epibenthos during the Quarter 3 IBTS-survey in the North Sea. It is constructed from galvanised steel. It was fitted with a 20 mm mesh and a liner of 4mm knotless mesh was fitted inside the codend. A chainmat was attached to protect the gear and to prevent the catch of heavy stones. It is fully described by Jennings *et al.* (1999). On the beam a net sonde was fixed to measure the distance over ground and to determine the exact point in time when the gear touched and left the bottom.

Triple beam trawl: Three standard 2m-beam trawls as described above were tied one behind the other by steel ropes of 6m in length on both sides (Figure 6.2.1.2). A net sonde was fixed on the beam of the first net.

To compare the catch rates of the gears the swept area of each haul was calculated by multiplying the towed distance over ground (satellite positions) by the effective width of the gear. The distance between the tips of the wings of both otter trawls were taken as the effective width of the trawls and for the beam trawls the distance between the beam trawl shoes. The catch data of each haul were then standardised to an area of one nautical square mile (nm^2). Only in the third experiment were catch data standardised to 500 m^2 .

Table 6.2.1.1. Data on fishing experiments.

vessel	gear	date	Box	no. of hauls	towing time (min)	mean effective swept area (m^3)
Solea	7m-beam trawl	12.-18.Dec.01	A	27	30	22766
Solea	Cod trawl	12.-18.Dec.01	A	27	30	71141
W. Herwig III	GOV	04.-07.Jan.02	A	19	30	79238
W. Herwig III	2m-beam trawl	04.-07.Jan.02	A	9	5	616
W. Herwig III	GOV	14.-16.Aug.03	A	20	30	88701
Solea	7m-beam trawl	16.-17.Aug.03	A	22	15	13014
W. Herwig III	2m-beam trawl	06.-10.Jan.04	A	13	5	460
W. Herwig III	triple beam trawl	07.Jan.04	A	3	5	636
W. Herwig III	2m-beam trawl	11.-12.Jan.04	N	8	5	459
W. Herwig III	triple beam trawl	10.Jan.04	N	3	5	612

6.2.1.3 Results

First experiment:

In Table 6.2.1.2 the mean catch rate standardised to 1nm^2 and its 95% confidence interval are listed for the 4 gears and all species caught. Mean catch rates with confidence limits $>100\%$ should be neglected, for these species were only caught in one or two hauls and in very low numbers.

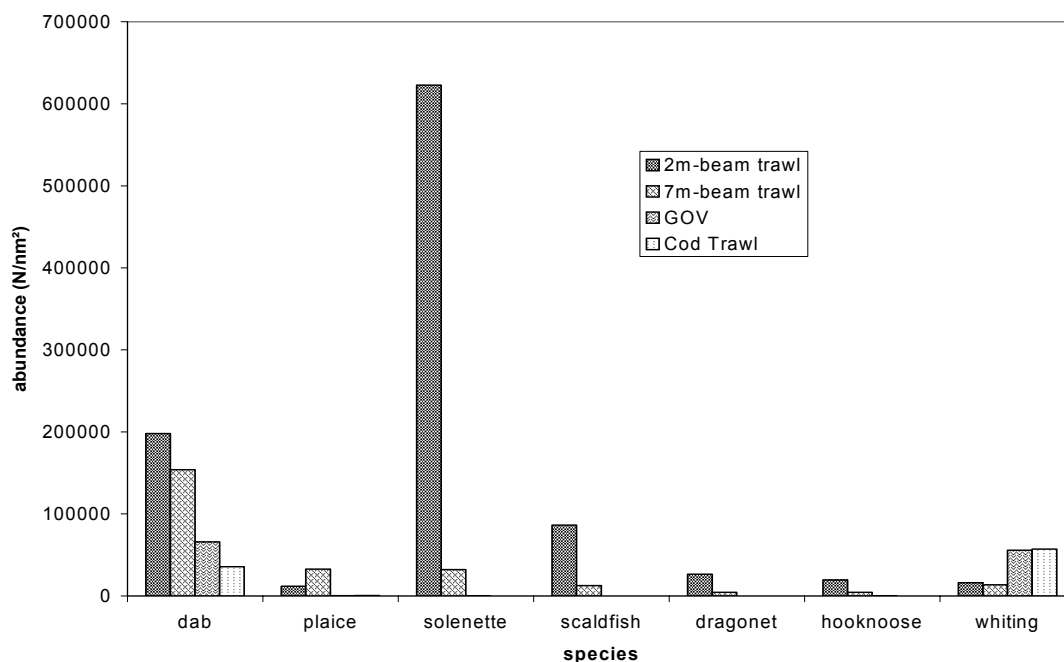


Figure 6.2.1.3. Differences in abundance estimates for several fish species in Box A (German Bight) based on catches of 4 different gears.

The highest estimates of abundance for most of the near bottom living or benthic species of small body size were obtained using the 2m-beam trawl. For solenette *Buglossidium luteum* the abundance was nearly 20 times higher compared to the 7m-beam trawl. The GOV only catches solenette randomly and this small flatfish was not recorded in the cod trawl. The same ranking also existed for scaldfish *Arnoglossus laterna*, dragonet *Callionymus* sp., hooknose *Agonus cataphractus* and dab *Limanda limanda*, whereas the 7m-beam trawl was the most effective gear for larger-bodied demersal species, such as larger plaice *Pleuronectes platessa*. With respect to whiting, the ranking was in the reverse order, and the abundance based on the Cod trawl and GOV was 3 to 4 times higher than that of the beam trawls.

Second experiment:

In summer 2003 the GOV was directly compared with the 7m-beam trawl in Box A. As shown in Figure 6.2.1.4 the proportions of bottom and pelagic species in both nets were naturally very similar compared to the results in the first experiment. The standardised mean catch and the 95% confidence interval are listed in Table 6.2.1.3.

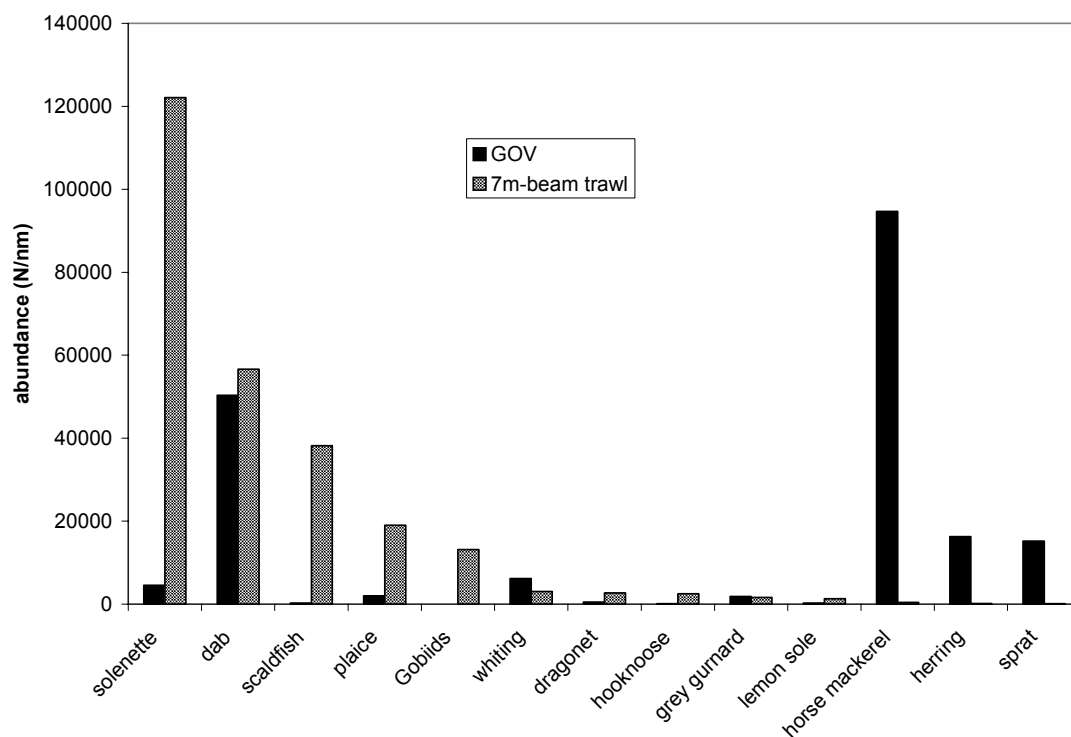


Figure 6.2.1.4. Differences in abundance estimates of fish species in Box A (German Bight) in summer based on catches of the GOV and the 7 m-beam trawl.

Table 6.2.1.2. Box A. Winter 2002. Comparison between 4 gears. Catch data are standardised to 1 nm².

species	GOV			KJN			7m-beam trawl		
	hauls	standardised mean catch	confidence interval (%)	hauls	standardised mean catch	confidence interval (%)	hauls	standardised mean catch	confidence interval (%)
BUGLOSSIDIUM LUTEUM	19	327	27.2	27			27	32234	14.0
AGONUS CATAPHRACTUS	19	123	44.1	27	15	81.6	27	4451	19.6
CALLIONYMUS LYRA	19	103	53.7	27	20	59.7	27	4591	23.8
CALLIONYMUS RETICULATUS	19			27			27		
CLUPEA HARENGUS	19	397008	64.9	27	4972	31.8	27		
ALOSA SPP.	19	5	134.7	27	5	142.1	27		
ECHIICTHYS VIPERA	19			27	2	196.0	27		
ENGRAULIS ENCRASICOLUS	19	9	89.8	27	2	196.0	27		
EUTRIGLA GURNARDUS	19	12	77.6	27	61	42.0	27	989	26.3
GADUS MORHUA	19	82	41.2	27	855	21.0	27	710	27.1
GOBIIDAE (INDET.)	19			27	5	109.0	27	610	32.3
AMMODYTES MARINUS	19	6	196.0	27			27		
HYPEROPLUS LANCEOLATUS	19	7	106.8	27			27		
HIPOGLOSSOIDES PLATESSOIDES	19			27	5	147.2	27		
LIMANDA LIMANDA	19	65869	35.9	27	35699	37.9	27	153858	21.9
LIPARIS SP.	19			27			27	5	196.0
MERLANGIUS MERLANGUS	19	55816	35.1	27	57096	20.3	27	13768	25.4
MICROSTOMUS KITT	19	5	196.0	27	12	75.1	27	44	70.2
MULLUS SURMULETUS	19			27	3	135.9	27	27	80.9
MYOXOCEPHALUS SCORPIUS	19	42	49.2	27	19	86.9	27	269	31.4
PETROMYZONIDAE	19	2	196.0	27			27		
PLATICTHYS FLESUS	19	61	55.0	27	27	47.6	27	12	136.2
PLEURONECTES PLATESSA	19	432	20.5	27	713	44.7	27	32587	15.6
POLLACHIUS POLLACHIUS	19	2	196.0	27			27		
POMATOSCHISTUS MINUTUS	19	12	97.6	27			27		
PSETTA MAXIMA	19	7	106.9	27	13	78.8	27	24	92.3
ENCHELYOPUS CIMBRIUS	19	10	89.5	27	5	108.8	27	36	116.8
SCOMBER SCOMBRUS	19	4	134.7	27	39	72.0	27		
ARNOGLOSSUS LATERNA	19	95	34.7	27			27	12700	10.9
SCOPHTHALMUS RHOMBUS	19			27	2	196.0	27		
SCYLIORHINUS CANICULA	19			27	2	196.0	27		
SOLEA SOLEA	19			27	2	196.0	27	46	59.5
SPRATTUS SPRATTUS	19	456747	55.4	27	1312	35.6	27	39	105.5
SYNGNATHUS ROSTELLATUS	19	5	134.7	27			27	69	67.6
TRACHURUS TRACHURUS	19	75	37.1	27	438	32.6	27	5	196.0
TRIGLA LUCERNA	19			27	5	196.0	27		
TRISOPTERUS LUSCUS	19			27	16	78.5	27	37	85.2
TRISOPTERUS MINUTUS	19	12	77.5	27	15	70.4	27	12	137.0

Table 6.2.1.3. Box A. Summer 2003. Comparison between GOV and 7m-beam trawl. Catch data are standardised to 1 nm².

species	GOV		confidence interval (%)	hauls	7m-beam trawl		confidence interval (%)	factor
	hauls	standardised mean catch			standardised mean catch	hauls		
BUGLOSSIDIUM LUTEUM	20	4554	45.0	22	122134		9.9	0.037
AGONUS CATAPHRACTUS	20	103	52.0	22	2483		19.8	0.041
CALLIONYMUS LYRA	20	480	20.7	22	2714		13.6	0.177
CLUPEA HARENGUS	20	16253	72.7	22	174		113.7	93.673
ALOSA SPP.	20	16	55.4	22				-
ECHIICHTHYS VIPERA	20	2	196.0	22				-
EUTRIGLA GURNARDUS	20	1848	41.8	22	1610		24.9	1.148
GOBIIDAE	20			22	13125		29.6	-
GADUS MORHUA	20	2	196.0	22				-
LAMPETRA FLUVIATILIS	20	17	98.3	22				-
LIMANDA LIMANDA	20	50339	15.9	22	56643		9.5	0.889
MELANOGRAMMUS AEGLEFINUS	20	2	196.0	22				-
MERLANGIUS MERLANGUS	20	6152	41.7	22	3063		42.0	2.009
MICROSTOMUS KITT	20	270	25.5	22	1281		21.1	0.210
MULLUS SURMULETUS	20	2	196.0	22				-
MYOXOCEPHALUS SCORPIUS	20	35	55.2	22	200		40.6	0.176
PHRYNORHOMBUS NORVEGICUS	20	2	196.0	22	23		135.4	0.086
PLATICHTHYS FLESUS	20			22	11		196.0	-
PLEURONECTES PLATESSA	20	1993	38.8	22	19018		9.5	0.105
POMATOSCHISTUS MINUTUS	20	316	72.5	22	12		196.0	26.618
PSETTA MAXIMA	20	21	59.6	22	181		51.3	0.116
ENCHELYOPUS CIMBRIUS	20	75	53.4	22	382		35.9	0.196
ARNOGLOSSUS LATERNA	20	272	31.0	22	38196		10.1	0.007
SCOMBER SCOMBRUS	20	421	58.7	22				-
SCOPHTHALMUS RHOMBUS	20	22	47.2	22	120		54.7	0.186
SOLEA SOLEA	20	8	90.0	22	616		29.7	0.013
SPRATTUS SPRATTUS	20	15162	62.3	22	84		75.7	179.897
SYNGNATHUS ROSTELLATUS	20	2	196.0	22	12		196.0	0.168
TAURULUS BUBALIS	20			22	12		196.0	-
TRACHURUS TRACHURUS	20	94687	43.9	22	379		56.3	249.566
TRIGLA LUCERNA	20	85	30.8	22	324		34.9	0.262
TRISOPTERUS LUSCUS	20			22	48		196.0	-
TRISOPTERUS MINUTUS	20	6	143.6	22				-

Third experiment

The third experiment was carried out aboard W. Herwig III in January 2004 in the Boxes A and N within the German Bight. It might be expected that catches would decrease from the first to the third net, and the distribution of the total catch in the three nets for various species are shown in Figures 6.2.1.5–8. Solenette, a small flatfish of 4–13cm length lives buried in muddy sand. It is possible that the first gear disturbs and digs out the specimens and they are then more vulnerable to the second and third nets. On average, only 26% of the total catch (by number) of solenette was caught in the first gear (Figure 6.2.1.5 and Table 6.2.1.4). The sculdfish, another small flatfish, showed a similar pattern (31%; Figure 6.2.1.6). The higher agility of species like sand goby *Pomatoschistus minutus* and dragonets *Callionymus* spp. could explain the higher proportion of catch in the first gear (52% and 37%; Figures 6.2.1.6–7 and Table 6.2.1. 4).

The mean catch of the single 2m-beam trawl hauls within the standard programme and the confidence interval were included in the figures (vertical lines). It shows that the catch of the single gear is of the same order than the catch of the first gear of the triple beam trawl. That means the two gears behind the first one do not affect the catch efficiency or catching characteristics of the first one (e.g., penetration depth of the chainmat). Therefore the results of the first net of the triple gear can be transferred to the single gear.

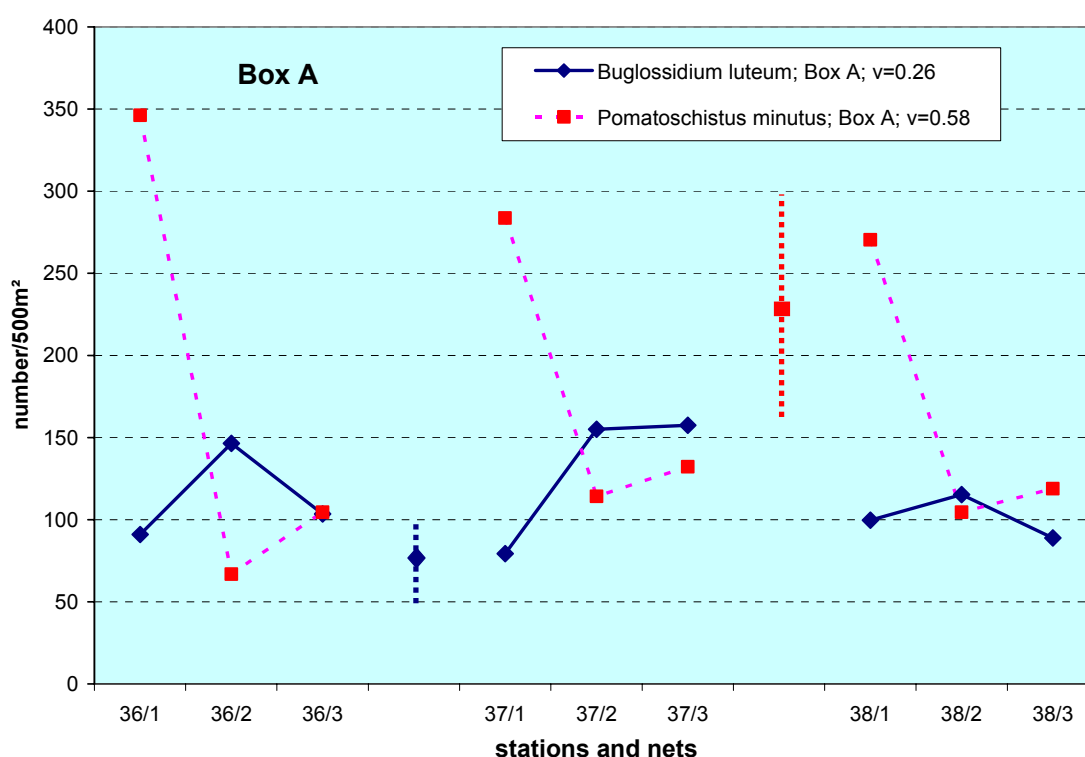


Figure 6.2.1.5. Catch of solenette (*B. luteum*) and sand goby (*P. minutus*) in the triple 2 m-beam trawl on 3 stations in Box A. v = vulnerability to the first gear. The vertical line represents the mean catch and the confidence interval of the single 2 m-beam trawl in the Box.

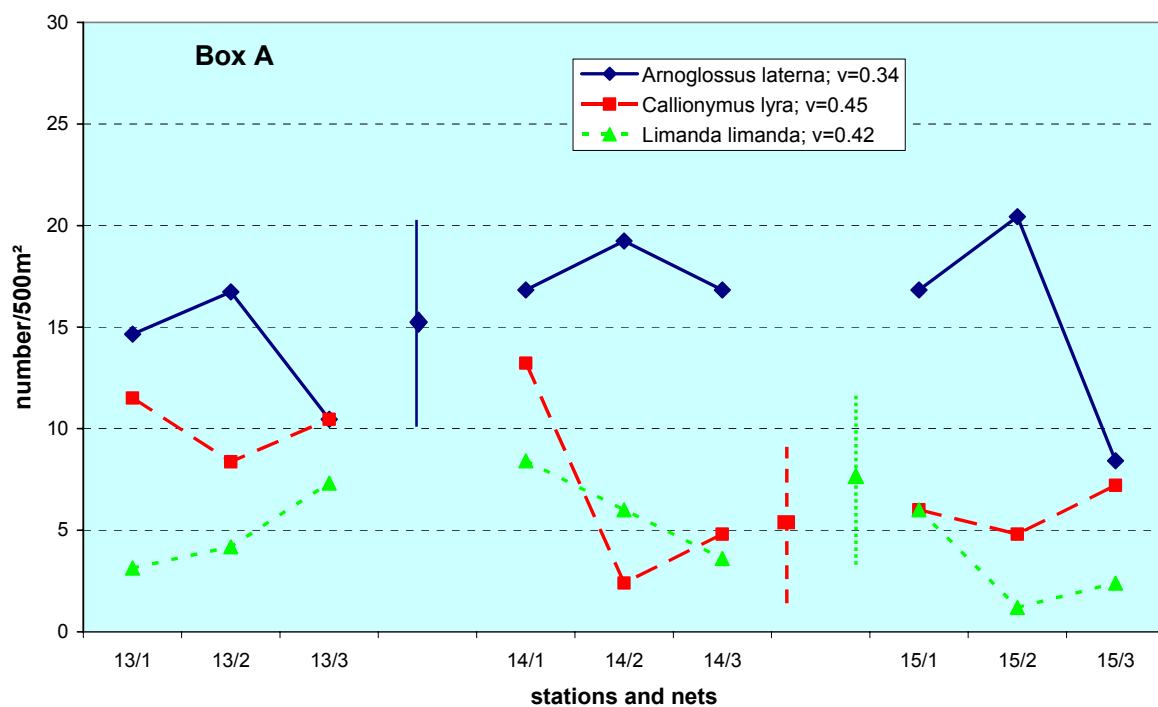


Figure 6.2.1.6. Catch of scaldfish (*A. laterna*), dragonet (*C. lyra*) and dab (*L. limanda*) in the triple 2 m-beam trawl on 3 stations in Box A. v = vulnerability to the first gear. The vertical line represents the mean catch and the confidence interval of the single 2 m-beam trawl in the Box.

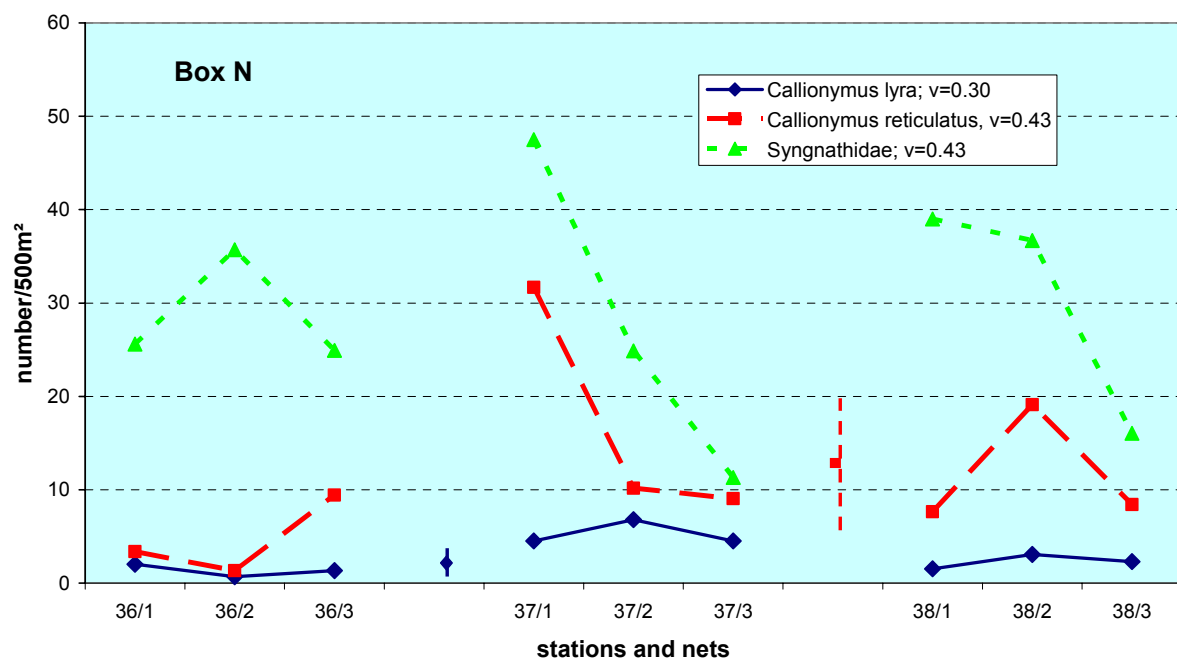


Figure 6.2.1.7. Catch of two dragonet species (*C. lyra* and *C. reticulatus*) and pipefish (*Syngnathidae*) in the triple 2 m-beam trawl on 3 stations in Box N. v = vulnerability to the first gear. The vertical line represents the mean catch and the confidence interval of the single 2m-beam trawl in the Box.

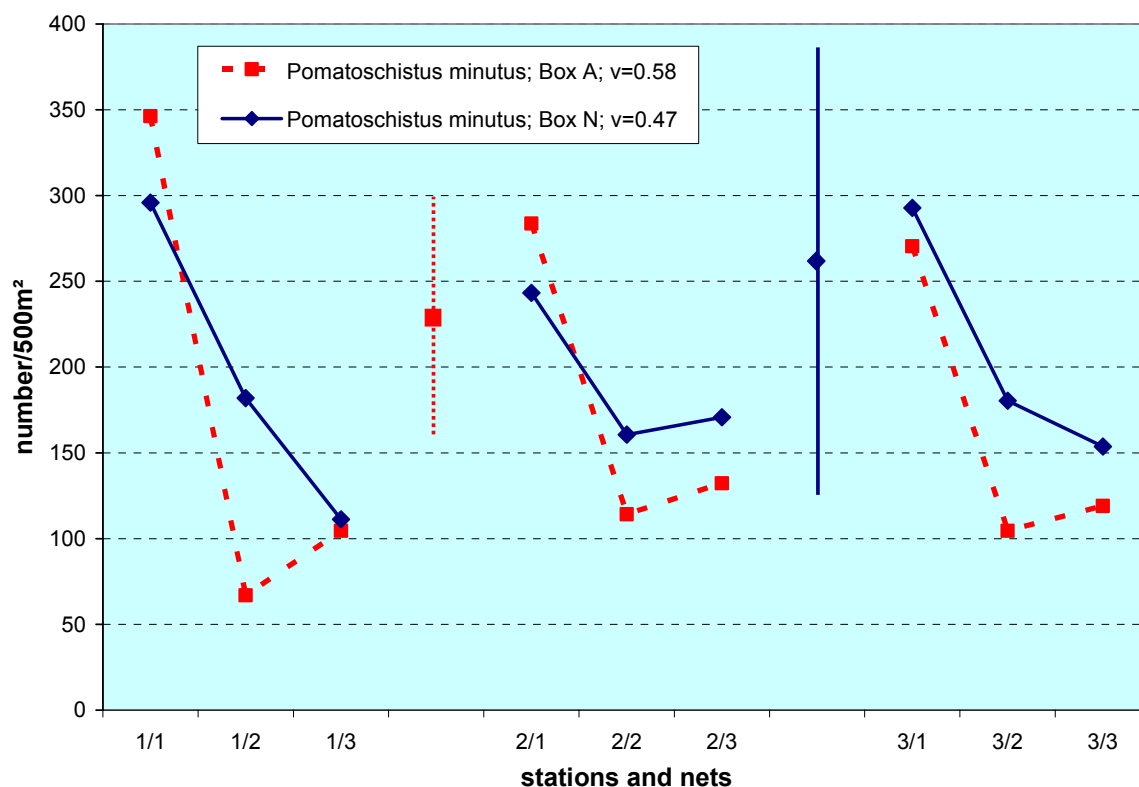


Figure 6.2.1.8. Comparison of the catch of sand goby in the triple 2 m-beam trawl on 3 stations in Boxes A and N respectively. v = vulnerability to the first gear. The vertical line represents the mean catch and the confidence interval of the single 2 m-beam trawl in the Boxes.

6.2.1.4 Vulnerability of fish species to the standard GOV

In Table 6.2.1.4 the factors between the gears (first and second experiment) and of the first gear of the triple 2m-beam trawl (third experiment) are listed. It is limited to species with confidence intervals of the mean <100%. Factors <1 means that the GOV is not so efficient in relation to the other gear. There are more specimens of this species in the path of the gear and the GOV only catch a part of them. Therefore the vulnerability of this species related to the GOV is smaller than 1.

The last row of Tab. 6.2.1.4 gives the vulnerability of the species to the standard GOV. Factors between the GOV and another gear of >1 result in a vulnerability of 1 according to the definition. From the first and second experiments the lowest value per species was taken. If more than one value per gear combination was available the mean was calculated. If the lowest factor was found in the quotient GOV/2m-beam trawl then this value was multiplied with the factor derived from the third experiment, assuming that the triple beam trawl catches the fish completely.

If a species like sole was only caught randomly by the single or triple 2m-beam trawl a correction factor from the third experiment could not be calculated. For plaice, the vulnerability index (given in Table 6.2.1.4) was a similar order of magnitude to sole. Looking at the results of the second experiment the factor for sole is only one tenth of the factor for plaice, and this relationship seems realistic from practical experience. Taking this into account the vulnerability index for sole should be estimated at a value of 0.001 or lower. The differences in the vulnerability between the species are illustrated in Figure 6.2.1.9.

Table 6.2.1.4. Factors between the gears (1st and 2nd experiment) and of the first gear of the triple 2 m-beam trawl (3rd experiment); limited to species with confidence intervals of the mean <100%. Vulnerability of the species to the standard GOV.

species	experiment 1 GOV/KJN	experiment 1 GOV/ BT7m	experiment 2 GOV/ BT7m	experiment 1 GOV/BT2m	experiment 3 1st gear/triple trawl	vulnerability GOV
TRISOPTERUS MINUTUS; poor cod	0.78120	-	-	-	-	0.78120
MERLANGIUS MERLANGUS; whiting	0.97757	4.05404	2.00885	3.47933	-	0.97757
CLUPEA HARENGUS; herring	79.85051	-	-	-	-	1.00000
MYOXOCEPHALUS SCORPIUS; bullrout	2.26146	0.15709	0.17551	-	-	0.15709
SPRATTUS SPRATTUS; sprat	348.07765	-	179.89678	-	-	1.00000
PSETTA MAXIMA; turbot	-	-	0.11637	-	-	0.11637
MICROSTOMUS KITT; lemon sole	-	-	0.21049	-	-	0.21049
EUTRIGLA GURNARDUS; grey gurnard	-	-	1.14816	-	-	1.00000
ENCHELYOPUS CIMBRIUS; four-bearded rockling	-	-	0.19632	-	-	0.19632
TRIGLA LUCERNA; tub gurnard	-	-	0.26249	-	-	0.26249
SCOPHTHALMUS RHOMBUS; brill	-	-	0.18619	-	-	0.18619
SOLEA SOLEA; sole	-	-	0.01294	-	-	0.01294
TRACHURUS TRACHURUS; horse mackerel	-	-	249.56559	-	-	1.00000
PLEURONECTES PLATESSA; plaice	0.60557	0.01325	0.10480	0.03611	0.24289	0.00877
ARNOGLOSSUS LATERNA; scadfish	-	0.00745	0.00713	0.00110	0.30843	0.00034
POMATOSCHISTUS MINUTUS; sand goby	-	-	-	0.00009	0.52000	0.00005
LIMANDA LIMANDA; dab	1.84512	0.42811	0.88870	0.33242	0.34289	0.11398
BUGLOSSIDIUM LUTEUM; solenette	-	0.01013	0.03728	0.00052	0.26047	0.00014
AGONUS CATAPHRACTUS; hooknose	8.03056	0.02757	0.04139	0.00630	0.18353	0.00116
CALLIONYMUS LYRA; dragonet	5.27144	0.02246	0.17684	0.00391	0.37427	0.00146

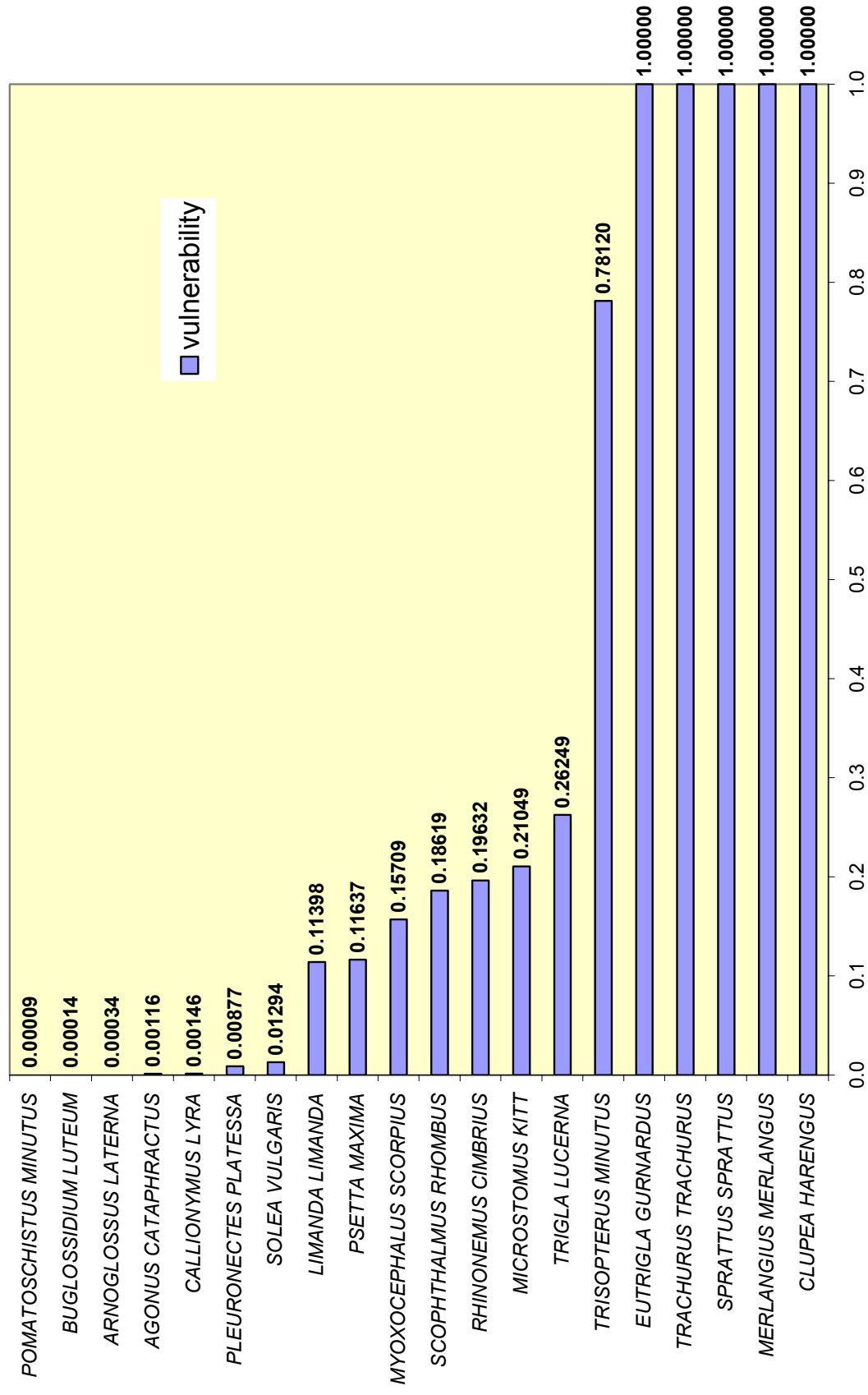


Figure 6.2.1.9. Vulnerability index of fish species related to the standard GOV

6.2.1.5 Discussion

The catches of a bottom trawl like the GOV contained a large variety of species; from small flatfishes (e.g., sole) to pelagic species living normally in the upper layer of the water column. On one hand, the contact of rubber disk ground ropes of the gear with the bottom is weak and individuals living in or very close to the bottom like solenette *Buglossidium luteum*, hooknose *Agonus cataphractus* and dragonets (*Callionymidae*) are underrepresented in the catches. On the other hand bottom trawls catch pelagic fish that are occasionally within the path of the net. Another characteristic of the GOV is the small-meshed liner of 20 mm mesh opening in the cod-end to prevent escapement of small fish through the meshes. Thus, the qualitative and quantitative species composition and the length composition in the catch from an area are gear specific and might not be representative of the fish fauna living in that area. Therefore, when describing a fish assemblage by using data from a single gear, the caveat that the description is gear-dependent should be highlighted.

The factors between the gears and the vulnerability indices given in this paper should not be used as conversion factors between the gears. The vulnerability of a species not only depends on gear parameters which can be kept constant, but is also dependent on environmental parameters like sediment structure and on the characteristics of the species like length composition, physiological condition and differences in the behaviour of age or length groups during fishing. All these parameters will change with the density of the species in front of the gear and perhaps with the time of the year.

A comparison of fish abundance data or only a list of them from different time periods not taken by the same gear can be misleading. After the collapse of the cod stocks on the Labrador Shelf, the survey gear was changed from a gear targeting cod to a gear targeting shrimps. It was impossible for them to find realistic conversion factors from the old to the new survey gear to avoid a break in the time series (Rätz, pers. comm.). Rijnsdorp *et al.* (1996) compared catch rates of demersal trawl surveys carried out in the period 1990–1995 and 1906–1909. For solenette, the standardised catch rate was 457.6 in the beginning and 0.5 at the end of the last century, caught by an imprecisely described otter trawl (OT20) and the GOV respectively. In a preliminary version of the North Sea Quality Status Report these two figures were compared by other authors and it was stated that not only the target species but also the by-catch species have decreased; for example solenette by a factor of 900. Taking the vulnerability index for solenette of 0.00014 into account a value of 0.5 means that at least 3500 individuals in the path of the gear were not caught by the GOV. This example may show how difficult, and in some cases impossible, it is to compare abundance indices derived from different gears. On the other hand this example and the results of the comparison experiments emphasize the importance of gear standardization and the validity of the basic rule: Never change the survey gear without having very serious reasons for that.

6.2.2 Case study 2: Analysis of relative catch efficiencies of selected fish species in survey gears

This text is adapted from the working document submitted by Niels Daan.

6.2.2.1 Introduction

One of the problems with the interpretation of the results of community analyses based on routine surveys is that catch rates of individual species are influenced by gear characteristics. Thus, different gears may show marked contrasts in species compositions and community metrics based on different surveys in the same area may deviate accordingly (Trenkel *et al.*, in press). The effect is that changes in the metrics tell us something about the sampled community rather than the 'true' community and any conclusion is conditional on the gear used.

To investigate the ecosystem effects of fishing, it would be ideal to be able to correct for differences in catch efficiency among species and gears. In another context, this problem has been approached in the past by identifying groups of species that might be considered to behave similarly to the same gear and to estimate the relative catchability of these groups based on representative exploited species for which absolute biomass estimates are available (Yang, 1982; Sparholt, 1987). However, little research has been done to underpin these groupings with quantitative information based on experimental evidence.

Information on the relative catch efficiency is also important from the perspective of integrating information from various surveys to obtain a coherent picture of the spatial and temporal distribution of individual species based on all information available, for instance with reference to the production of a fish atlas. These considerations provide the background of TOR e of the April 2004 meeting of the Working Group on Fish Ecology, to "*start analyses of relative catchabilities of the more common fish species in different survey gears*". This analysis provides some exploratory results of a comparison of the catch rates of some of the more abundant fish species in five North Sea surveys. Exploratory analyses from a comparison of the catch rates of some of the more abundant fish species in five North Sea surveys were made: the International Bottom Trawl Survey (IBTS), the Beam Trawl Survey (BTS), the Demersal Fish

Survey (DFS), the Sole Net survey (SNS) and the MAFCONS project. The primary aims were to investigate (1) whether the available data provide qualitatively consistent results and (2) whether the relative catch efficiency can be estimated quantitatively with sufficient confidence for practical use.

The comparison is entirely based on data that have not been collected specifically for comparative purposes. Each survey has its own constraints in terms of spatial and temporal distribution and overlap between stations has a haphazard distribution. Therefore, criteria must be set regarding the selection of 'comparable' hauls. For this purpose, the year was stratified in four quarters and an ICES rectangle in nine sub-rectangles of 10*10 nm. All comparisons refer to parallel hauls made with different gears within the same stratum.

6.2.2.2 Methods

The five surveys available for the comparison were:

- a) IBTS, 1977–2003 (incomplete data for 2001–2003; 1st Q all years; other quarters variable; standard haul duration: 30 min) – gears investigated: GOV (Chalut à Grande Ouverture Verticale), ABD (Aberdeen trawl - 3rd Q 1992–1996 - and Granton trawl - 3rd Q 1991 - combined), DHT (Dutch Herring Trawl in various riggings including the High-Opening Bottom Trawl). Fishing speed 4 knots.
- b) BTS, 1985–2003 (3rd Q; standard haul duration: 30 min) – gear: 8m-beamtrawl with chains (+c) and with (+s) or without stone barrier (-s). Fishing speed 4 knots.
- c) DFS, 1977–2003 (various quarters; standard haul duration: 15 min) – gear: 6m-beamtrawl with wooden bobbins (+b). Fishing speed 3 knots.
- d) SNS, 1977–2003 (various quarters; standard haul duration: 15 min) – gear: 6m-beamtrawl with chains (+c). Fishing speed 3 knots.
- e) MAFCONS, 2002–2003 (3rd Q; standard haul duration: 5 min) – gear 2m beam trawl with chain mat (+m). Fishing speed 1 knot.

All catches were raised to 30 min hauls, but no effort was made at this stage to take into account swept areas based on the width of the fishing gear and/or fishing speeds. This may have implications for the results for some species (e.g., gurnards, where CPUE in beam trawls and GOV trawls appear similar), although the general rules will be consistent.

All hauls were assigned to an ICES sub-rectangle (20' longitude and 10' latitude: 9 per rectangle) according to the shooting position. Parallel hauls made with two or more gears made within one sub-rectangle and within the same year and quarter were selected from the database. In case more than one haul was made with one of the gears in a paired set, the average catch was taken when making the comparison to avoid double counting in estimating catch ratios. The comparisons made, the number of paired sets, numbers of hauls, the number of 'positive paired sets' per species (number of sets in which a species has been recorded in at least one of the gears) and the number of positive hauls by species are given in Table 6.2.2.1. Zero sets were excluded from the comparisons for individual species, because these were considered to carry no information on catch efficiency ('structural zeros'). Thus, the amount of data available for individual species varies widely by set depending on absolute abundance and distribution range.

Some species such as the larger ray species and dragonets were combined because of identification problems in the database (Daan, 2001). Rare species were omitted entirely.

Two types of analyses were performed: (1) To obtain a global view of the relative catch efficiency by species for each gear comparison, the catch ratio by positive paired set was ln-transformed to normalise relative catch rates, after adding an arbitrary value of 0.01. (2) In an effort to correlate the catch rates of the two gears and to quantify the relative catch efficiency for a specific comparison of the GOV and the 8m beam trawl, for which a large number of sets were available, the catch rates themselves were log-transformed after adding an arbitrary value of 0.1. The individual catch rates by species were then correlated and their frequency distribution investigated. In both cases the arbitrary values added were scaled to the minimum value in the data set, so as to have least influence to the data as a whole.

Data analyses were performed in SAS.

6.2.3 Results

Analysis of catch ratios

Figure 6.2.2.1a-d provides the results for all gear combinations investigated grouped by type of comparison. All species for which all hauls made by one of the gears were zero were excluded, because in this case the catch ratio is completely arbitrary.

The catch ratios of the three beam trawl gears relative to the GOV (Figure 6.2.2.1a) showed a remarkably consistent pattern, individual species often showing similar trends in different beam trawl surveys. With the exception of argentine and sandeels, pelagic species were highly underrepresented in the beam trawl, and to a lesser extent also gadoids, with the exception of poor cod *Trisopterus minutus*, bib *Trisopterus luscus*, ling *Molva molva* and rocklings. In beam trawls, other demersal fish were generally caught in relatively larger numbers per unit time, but grey gurnard *Eutrigla gurnardus* and snake blenny *Lumpenus lampretaeformis* represented notable exceptions and lesser weaver *Echiichthys vipera* were caught in equal numbers. Flatfish also revealed marked differences. Whereas scadfish, sole and solenette were poorly represented in the GOV, long rough dab *Hippoglossoides platessoides*, dab and lemon sole *Microstomus kitt* were caught in equal numbers. It should be noted that the number of paired hauls for the DFS/GOV comparison was limited (19) compared to the SNS/GOV (111) and BTS/GOV (316) comparisons.

The intra-IBTS comparisons of the three main gears (ABD/DHT/GOV; Figure 6.2.2.1b) used in the past indicated much smaller but also systematic differences. The DHT (78 pairs) generally had somewhat higher catch rates throughout the various species groups, while the ABD (72 pairs) had somewhat lower catch rates.

Among the different beam trawls used (Figure 6.2.2.1c), the DFS appeared more effective than the SNS (316 pairs) for pelagic species and less so for demersal species, with a few exceptions. The number of pairs for the DFS/BTS comparison yielded variable results, but the number of pairs was quite small (16). Also the SNS/BTS catch ratios were variable, even though the number of pairs was larger (80). The two types of gears used in the BTS, and the gear with a stone barrier was consistently lower for the three main commercial gadoids and the two main flatfish species, but there was little overlap in the use of these gears (15 pairs).

Finally, catch rates in the 2m-BT used for benthos sampling in recent years during the BTS survey (66 pairs) was consistently lower for most gadoids and flatfish, but there were some notable exceptions. Hagfish *Myxine glutinosa*, pipefish (*Syngnathidae*), sea snails *Liparis* spp., snake blenny, butterflyfish *Pholis gunnellus* and sand gobies, and to a lesser extent sandeels and solenettes, were caught in much higher numbers per 30 min haul, even though the swept area is much lower.

Regression analysis

Figure 6.2.2.2 a-d provides plots of mean $\log(n/hl)$ by year in the GOV against those in the BTS survey (8m-BT with and without stone barrier combined) by species for flatfish, roundfish, other demersal species and pelagic fishes separately. Only species that were represented systematically have been included. These clearly show that individual species represent often rounded (e.g., dab, plaice, grey gurnard) or elongated clusters (e.g., long rough dab, haddock, whiting). These clusters lie systematically below or above the diagonal at approximately equidistance, suggesting that catch efficiency is consistent in these cases among years. There are also notable exceptions (e.g., butterflyfish) and the pattern in pelagics was less clear because of the overriding number of zero catches in the BTS in many years.

Figure 6.2.2.3a-i illustrates the $\log(n/hl)$ values in the two gears by pair for nine selected species and their frequency distributions by class. Although the scatter is often considerable and may easily cover three orders of magnitude in absolute terms on both axes, the frequency distributions reveal clear and consistent differences for all species. The corresponding basic statistics for each species are given in Table 6.2.2.2. With the exception of herring, all correlations were significant at $p < 0.01$ and up to 59% of the total variance can be explained by the relationship (dab). Because the choice of the independent variable is completely arbitrary in this comparison, the slope of the functional regression rather than the slope of the common regression is of interest here. Again with the exception of herring, these slopes are close to 1 and probably not significant (there are problems in determining the formal significance for functional regressions).

6.2.3.1 Discussion

The exploratory data analysis presented serves merely to show that the comparison of catch rates in survey gears, even if collected haphazardly, leads to consistent differences among species that should reflect differences among individual

species in the catch efficiency. However, the most appropriate measure for the catch ratio, and its confidence limits, is not easily resolved. In the current study, it was considered that leaving out pairs for which one of the gears yielded a zero catch would cause a bias, because the zero catches are not equally distributed among both gears but dominate among the least efficient gear. However, by including zero catches one has to make an arbitrary choice about the value to add before the log transformation, which will affect the estimated average catch ratio.

The potential for estimating actual conversion factors among surveys obviously varies among species, depending on the abundance and distribution relative to the timing and area of overlap. However, given the high variability of catch rates within gears, the variance explained by the BTS/IBTS correlation for instance for whiting ($r^2=0.59$) and plaice ($r^2=0.50$) should ensure a fairly reliable estimate. One factor that has not been considered explicitly so far is the size range of the catch. It may well be that, within species, catch efficiency is strongly dependent on size and the explained variance might be even higher if this factor was taken into account. Other potential factors affecting catch efficiency relate to sediment type and depth, but of course degrees of freedom is rapidly decreasing when introducing multiple explanatory factors.

A species-wide analysis such as this should be able to give important guidance as to what species should be sensibly included in community analyses from specific gears. It is quite clear that most sharks and pelagics are so rarely caught in beam trawl that it is doubtful that such data would add anything except white noise. Also, based on the extremely high abundance of smaller species such as gobies and solenette in the (small meshed) 2m beam trawl, leads to the question whether the 8m beam trawl provides an adequate sample of these species in absolute or even relative terms.

In evaluating total North Sea fish biomass, Yang (1982) and Sparholt (1990) assumed that, with the exception of sole, all flatfish species had the same chance to be caught in Granton trawls. However, the results presented here indicate that relative catch efficiency in the 8m beam vs. GOV varies considerably among species. Whereas plaice, turbot and brill were clearly underrepresented in the GOV, dab, long rough dab and lemon sole appeared in the catch in approximately equal numbers in the two gears, suggesting that such grouping may not be entirely satisfactory. Assuming a catch efficiency of dab that is equal to the one for plaice may grossly overestimate its biomass (17% of the total fish biomass according to Sparholt (1990)).

It was not considered appropriate at this stage to estimate the approximate conversion rates, because other factors first need to be explored in subsequent analyses, before this could be done satisfactorily. Although the results presented must be considered as preliminary, a few conclusions may be drawn:

- 1) The available data appear to contain enough positive pairs (even at a scale of 10*10 nm and by year and quarter), at least for the more abundant species and for some gears, to investigate size-dependent catch ratios in detail.
- 2) Even within major groupings such as gadoids or flatfish relative catch efficiency of individual species may vary consistently.
- 3) Further analyses should reveal size-dependent differences in catch efficiency.
- 4) Even though accurate conversion rates may not always be possible, large discrepancies in catch rates provide objective information on species that are inadequately sampled by specific gears (e.g., pelagics in beam trawls; hagfish in all gears except for the 2m beam trawl).

Comparison		DFS	BTS	DFS	SNS	DFS	IBTS	SNS	IBTS	IBTS	IBTS	IBTS	SNS	BTS	BTS	IBTS	BTS	BTS	MAFCONS												
8857040603	Hippoglossoides platessoides							2	0	2	69	69	72	58	50	50	191	163	217	11	11	9	53	24	52						
8857040904	Limanda limanda	16	45	9	311	837	473	19	39	20	112	158	117	67	65	72	66	65	46	80	113	61	356	374	400	15	15	16	60	42	60
8857041202	Microstomus kitt	1	1	0	41	22	26	4	0	4	34	26	24	65	60	70	45	31	28	25	20	17	261	227	242	12	11	12	59	14	59
8857041402	Platichthys flesus	8	5	7	248	270	267	15	12	11	50	42	35	3	3	1	14	15	6	44	31	37	46	48	25	1	1	1	2	0	2
8857041502	Pleuronectes platessa	16	46	14	316	853	479	19	40	19	111	157	108	61	52	59	44	42	20	80	112	71	348	372	352	15	15	16	62	9	62
8858010601	Solea vulgaris	16	40	14	306	724	446	15	31	9	102	147	22	2	2	0	6	2	5	79	109	77	221	241	38	9	5	8	12	0	12
8858010801	Buglossidium luteum	16	25	9	232	368	229	8	6	4	104	134	44	4	3	2	4	4	0	72	88	58	208	214	116	11	10	9	18	17	16

Table 6.2.2.2. Summary statistics (log abundance+0.1) for individual data pairs by species for BTS (8m beamtrawl with and without stone barrier) and IBTS (GOV).

# observations		Cod	Whiting	Plaice	Sole	Dab	Grey gumard	Hooknose	Dragonets	Herring
		280	357	348	221	356	318	229	356	283
BTS	average	0.0177	1.0246	1.5785	0.8981	2.1819	0.9854	0.7377	0.9826	-0.8574
	median	0.0414	1.0453	1.5809	0.8855	2.3703	1.1790	0.7076	1.0828	-1.0000
	s.d.	0.8634	0.9417	0.7745	0.6732	0.8117	0.7699	0.8285	0.9351	0.4463
	s.e.	0.0516	0.0498	0.0415	0.0453	0.0430	0.0432	0.0547	0.0496	0.0265
IBTS	average	0.6377	2.3710	0.7623	-0.7596	2.2507	1.2891	-0.5673	0.3903	2.1746
	median	0.5824	2.4595	0.8799	-1.0000	2.4655	1.4330	-1.0000	0.3222	2.3245
	s.d.	0.9126	1.0871	0.8500	0.5925	0.8895	0.8655	0.7805	1.1644	1.4156
	s.e.	0.0545	0.0575	0.0456	0.0399	0.0471	0.0485	0.0516	0.0617	0.0841
correl		0.24	0.41	0.50	0.35	0.59	0.35	0.21	0.32	-0.02
significance		p<0.01	p<0.01	p<0.01	p<0.01	p<0.01	p<0.01	p<0.01	p<0.01	N.S.
slope		0.2614	0.4790	0.5474	0.3039	0.6428	0.3901	0.2060	0.3981	-0.0723
slope functional relationship		1.0783	1.1548	1.1014	0.8595	1.0947	1.1061	0.9618	1.2411	-3.2415
s.e. yx		0.8312	0.8220	0.6508	0.6410	0.6139	0.7036	0.8025	0.8618	0.4430

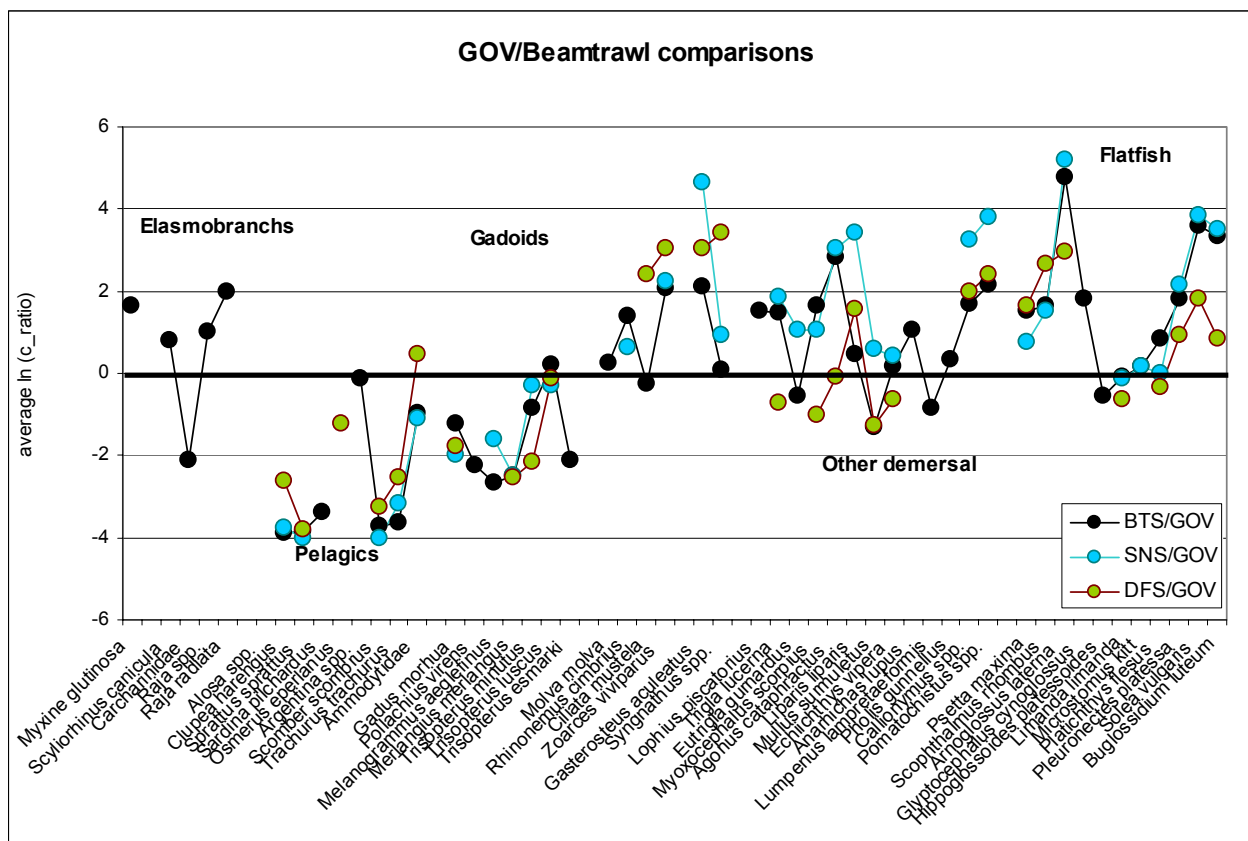


Fig. 6.2.2.1a. Average $\ln(\text{catch ratio} + 0.01)$ by species for inter-gear comparison of positive paired sets: a. various beam trawls vs. GOV.

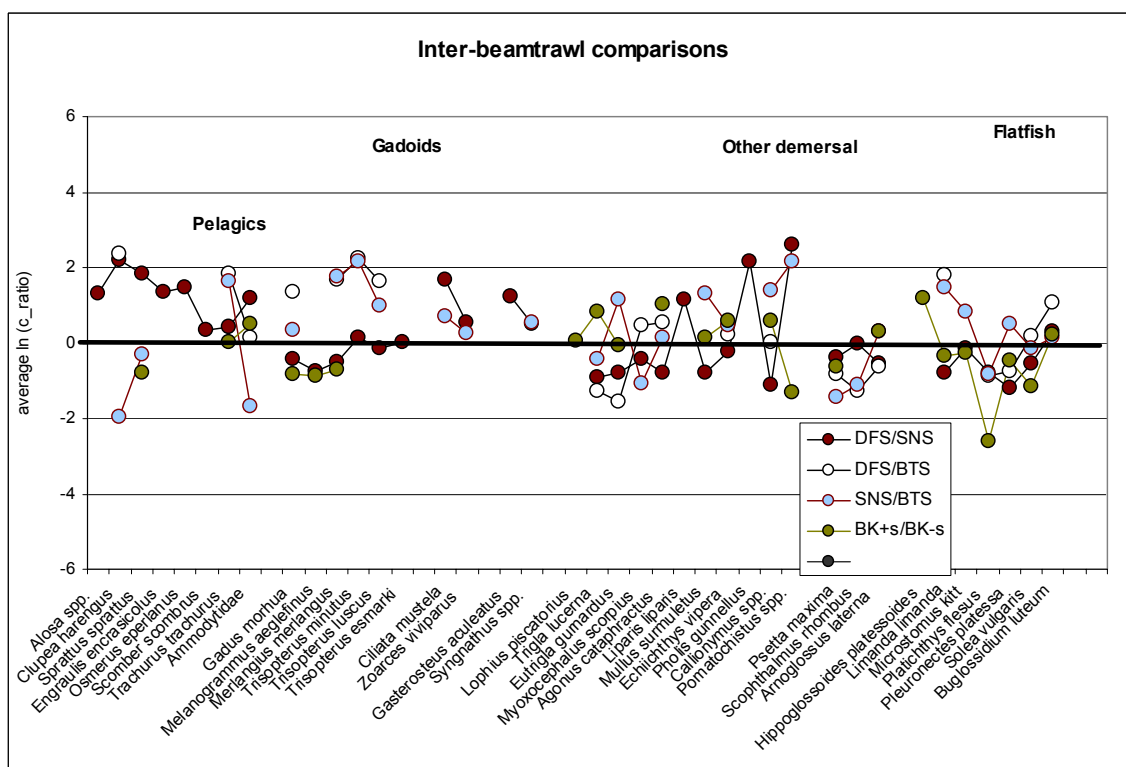


Figure 6.2.2.1b. Inter-beam trawls comparisons.

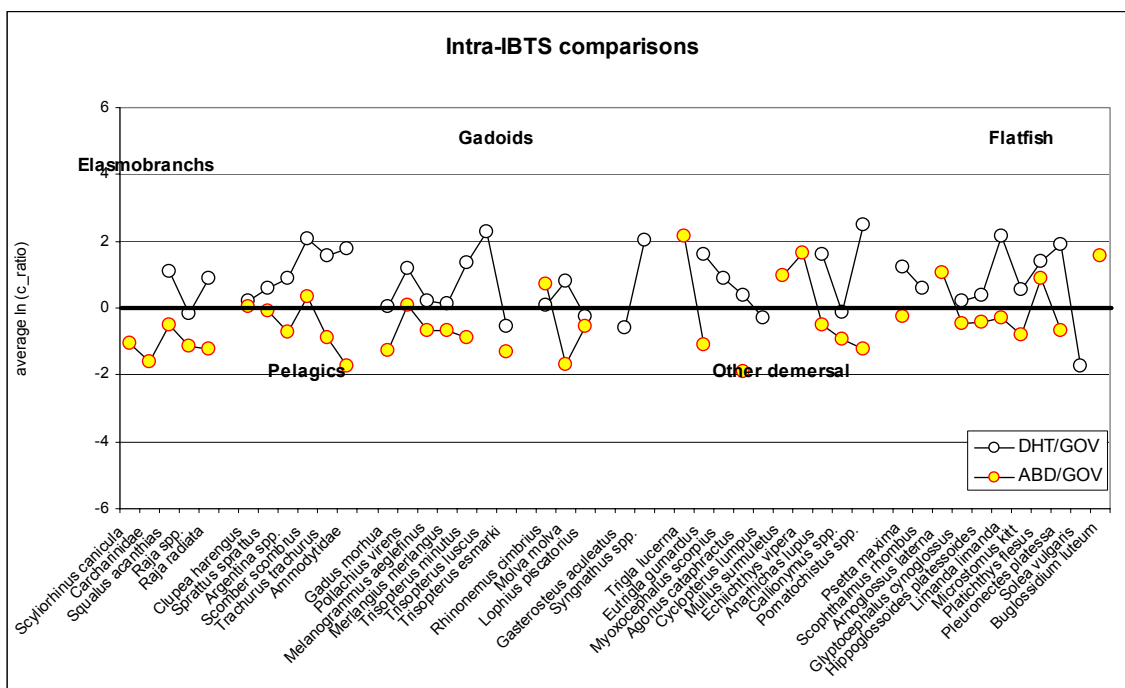


Figure 6.2.2.1c. DHT and ABD vs. GOV comparison.

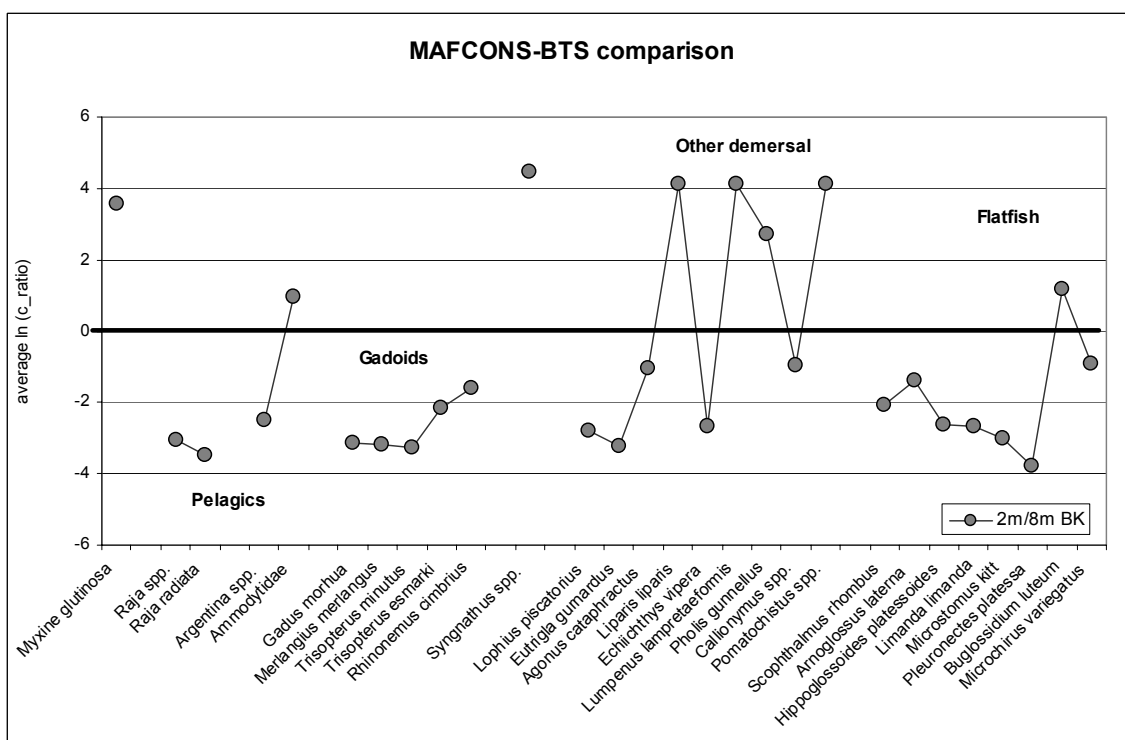


Figure 6.2.2.1d. 2m vs 8m beam trawl comparison.

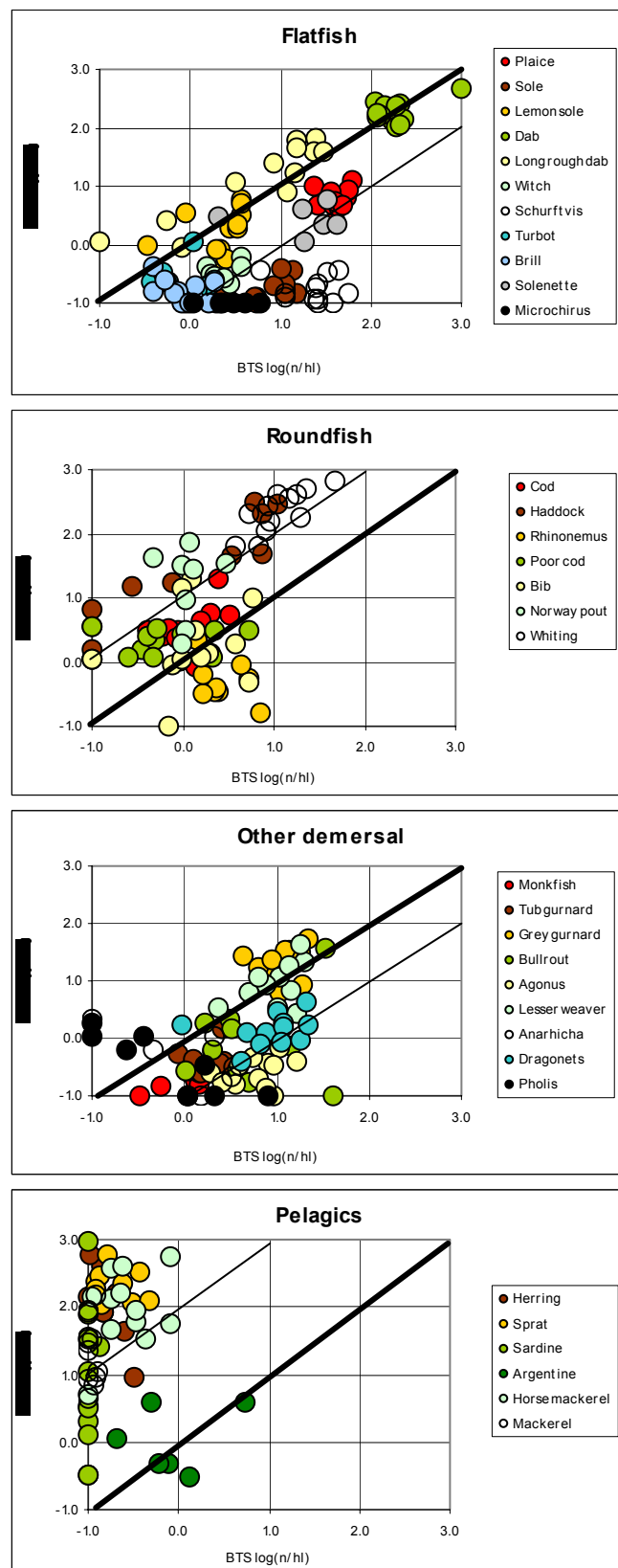


Figure 6.2.2.2. Average annual estimates of $\ln(\text{catch per haul})$ in GOV vs 8m beam trawl for different groupings: a. flatfish; b. gadoids; c. other demersal; d. pelagics.

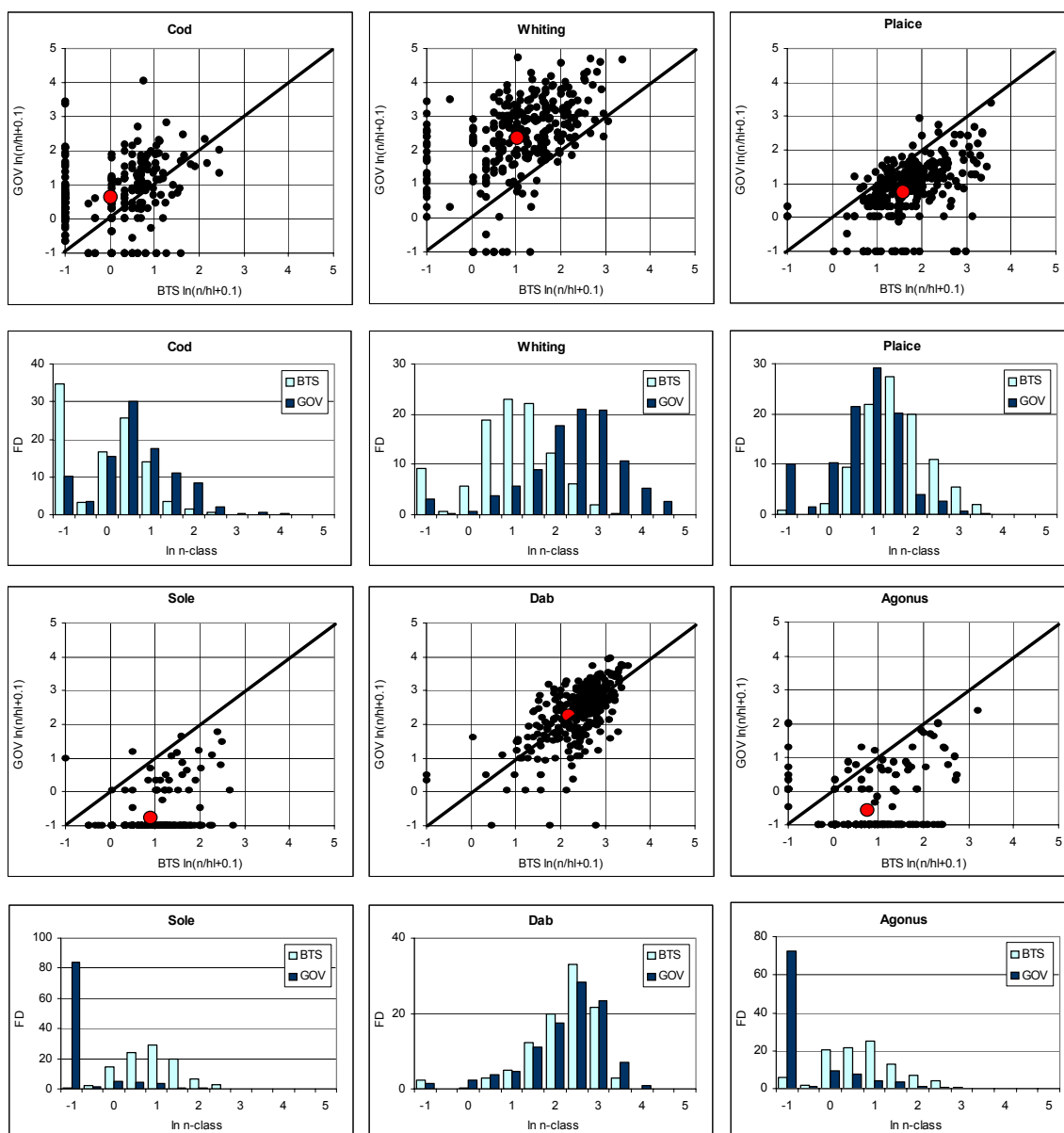


Figure 6.2.2.3. Plot of $\log(\text{abundance}+0.1)$ in GOV vs 8m beam trawl for paired sets and their frequency distributions by log class for selected species.

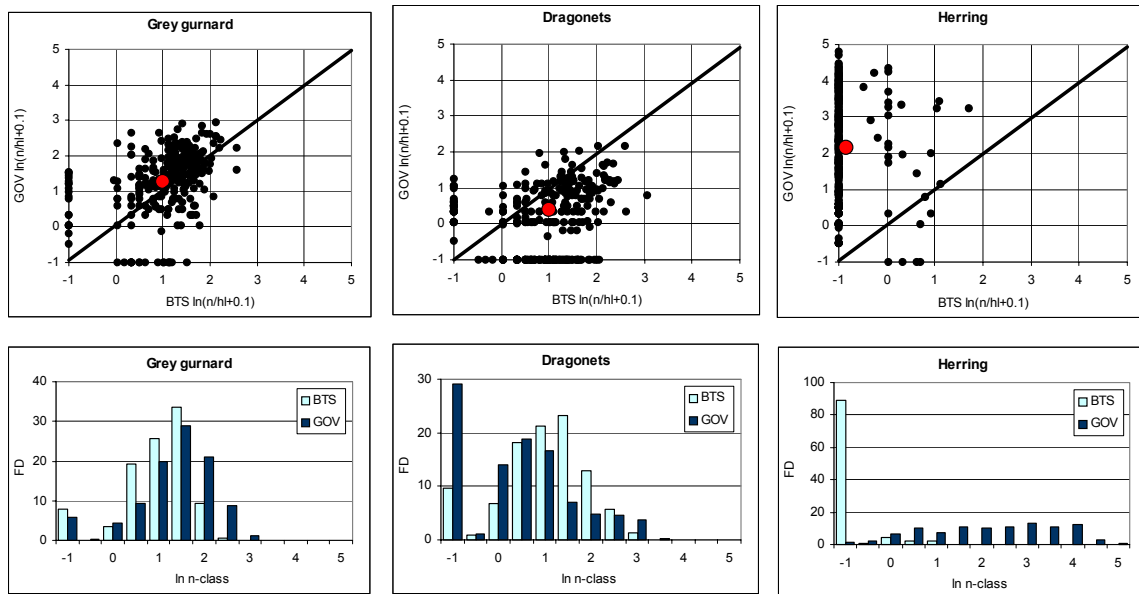


Figure 6.2.2.3 Continued. Plot of log (abundance+0.1) in GOV vs. 8 m beam trawl for paired sets and their frequency distributions by log class for selected species.

6.3 Conclusions

- Changing or modifying gears in long-term surveys will affect the catchability of the various species and sizes of fish, therefore hindering analyses of long-term trends. Hence, maintaining constant gear specifications and survey protocols is of major importance.
- Further studies on relative catchability are required, and can utilise existing data to determine general relationships. Field work may be required to examine more specific relationships (e.g., the relative catchabilities of gears on specific sediment types).
- Future data analyses should incorporate size-based analyses (e.g., size distributions of fishes between gears, differences in relative catchability by size-class of major fish species).
- The conversion of survey data from CPUE (numbers per hour) to relative abundance per swept area should be encouraged in such analyses, and in the case of otter-board type trawls, data on wing-spread should be collected.
- Even within similar taxa (e.g., flatfish), there can be major differences in the relative catchability. Better estimates of catchability for all species by substrate are required if more accurate estimates of fish biomass are to be made.
- For the purposes of examining regional fish community issues, the use of multiple gears is vital if data are to be collected for as many components of the fish fauna as possible. Spatially comprehensive surveys with 2m beam trawl (small-bodied demersal species); 4–8m beam trawl (large-bodied demersal species) and GOV trawl (large-bodied demersal and pelagic species) provide data for the majority of the fish assemblage. There are some taxa of fish (e.g., sand eels, epipelagic fish such as garfish and small-bodied pelagic fish) and ground types (e.g. very shallow water and rocky habitats), however, which are not effectively sampled by these surveys.

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7 STATUS AND CHANGES IN FISH SPECIES DISTRIBUTIONS AND FISH COMMUNITY COMPOSITION AND INTERACTIONS IN THE NORTH SEA (2000–2004).

7.1 Introduction

7.1.1 Background

At the Intermediate Ministerial Meeting (IMM) on fisheries in Bergen (1997), the Ministers of the North Sea countries agreed as one of the guiding principles: *Further integration of fisheries and environmental protection, conservation and management measures, drawing upon the development and application of an ecosystem approach which, as far as the best available scientific understanding and information permit, is based on in particular:*

- *the identification of processes in, and influences on, the ecosystems which are critical for maintaining their characteristic structure and functioning, productivity and biological diversity;*
- *taking into account the interaction among the different components in the food-webs of the ecosystems (multi-species approach) and other important ecosystem interactions; and*
- *providing for a chemical, physical and biological environment in these ecosystems consistent with a high level of protection of those critical ecosystem processes.*

Subsequently, a workshop on the Ecosystem Approach to the Management and Protection of the North Sea was held in Oslo (June 1998). This workshop identified monitoring as a key component of an ecosystem approach in relation to ecological objectives, assessments, and scientific advice to management.

The 5th North Sea Ministerial Conference met in Bergen (20–21 March 2002) with the aim of agreeing to implement an ecosystem approach, based on a conceptual framework developed at the 1998 workshop.

In 1997 an ICES Steering Group on the Global Ocean Observing System (SGGOOS) was formed in order to prepare an action plan as to how ICES should take an active and leading role in the further development and implementation of GOOS at a North Atlantic regional level, with special emphasis on operational fisheries oceanography. At a workshop convened in Bergen (1999) a draft design and implementation plan was conceived, with three essential components:

- To promote global / regional linkages in a GOOS context.
- To promote the ICES Annual Ocean Climate Status Summary as a contribution to GOOS.
- To design and implement a North Sea ecosystem component of GOOS in collaboration with EuroGOOS.

In order to develop these suggestions further, SGGOOS was renamed in 1999 as a joint ICES/IOC Steering Group on GOOS with the terms of reference to further develop an Implementation Plan. The SGGOOS initiated a workshop co-sponsored by IOC, ICES, OSPAR, the North Sea Conferences and EuroGOOS in September 2001 to agree on a strategy for a pilot North Sea Ecosystem GOOS project.

In order to meet the challenges identified at the meeting, the workshop agreed to increase the efficiency and effectiveness of the use of data products from current relevant national and international monitoring, and therefore invited the national agencies responsible for monitoring of the North Sea to:

- establish a co-ordinated mechanism that could add value to existing activities by integrating data from various sources (physical, chemical, biological) to aid development of an ecosystem approach,
- collaborate by means of a pilot project sponsored by ICES and EuroGOOS to demonstrate the usefulness of this approach by integrating data on oceanography and fisheries.

Although considerable progress has been made recently by a variety of national agencies and through EuroGOOS on monitoring, modelling, and forecasting physical parameters, until now no attempt has been made to establish an integrated information system for the North Sea which includes ecosystem parameters.

The present monitoring of the North Sea is generally insufficient to discriminate between human impacts and natural variation on the ecosystem. There is a need for improved, integrated monitoring through co-ordination and harmonisation of existing national and international monitoring activities, as well as through implementation of new methods and technology.

For marine ecosystems, meteorological and climatic variability are primary driving forces for ecosystem variability. Improved knowledge of the relationship between climate and changes in ecosystems would facilitate studies that aim to distinguish between anthropogenic impacts and natural variability in environmental assessments.

The North Sea, because of the intensive work that has already been carried out in this area, was an obvious candidate for a pilot project. Developing an ecosystem approach for the management of the North Sea requires coordinated, integrated monitoring and information systems, and a continuous updating of information.

7.1.2 North Sea Pilot Project (NORSEPP)

The NORSEPP project involves fishery scientists and oceanographers from all North Sea countries (ICES, 2003a). The utility of incorporating environmental/ oceanographic data into the assessments of certain species could be explored. Such an approach will involve the bringing together of very large data sets and the application of new approaches to fishery assessment modelling.

The overall objective is: To initiate operational fisheries oceanography by integrating existing physical, geochemical, and biological monitoring programmes and models to improve advice to fisheries managers.

There are seven specific objectives, each of them related to a specific work package:

- *Co-ordinate and harmonise relevant physical, geochemical and biological monitoring programmes and networks.*
- *Combine models and data to generate operational products relevant to fish stock assessment*
- *Further develop and demonstrate applications of operational products to fish stock assessment using test cases*
- *Streamline the flow and exchange of data and information*
- *Evaluate the outputs of the operational systems in order to assess their usefulness and accuracy, and suggest improvements where necessary*
- *Establish efficient communication in order to disseminate operational products to users*
- *Evaluate existing North Sea monitoring technologies and strategies regarding their usefulness for operational fisheries oceanography and initiate improvement*

The focus on living resources is intended to limit the scope of the project to something achievable within the time frame. If the project succeeds, its remit could be expanded to determine the usefulness of this approach as a tool for comprehensive environmental analysis in support of improved environmental assessments.

One of the specific analyses required by the NORSEPP is the visualisation and analysis of historic and present fish distribution (spawning, feeding and fishing areas) in relation to water mass distribution and characteristics (including oxygen conditions). To demonstrate to what degree the fish distribution is explained by the environmental conditions.

7.1.3 Regional Ecosystem Study Group for the North Sea (REGNS)

Following from work undertaken by NORSEPP, REGNS was tasked with “looking ahead beyond the North Sea Pilot Project at the future needs to support member countries in the delivery of the Ecosystem Approach” (ICES, 2003b). WGFE was asked in ToR f to “start preparations to summarise status and changes in fish species distribution and fish community composition and interactions in the North Sea for the period 2000–2004, for input to the Regional Ecosystem Study Group for the North Sea in 2006”.

7.2 Status and changes in fish species distribution and fish community composition and interactions in the North Sea for the period 2000–2004

7.2.1 Data availability

The main data that are available for examining the status and changes in fish species distribution and fish communities are comprised of survey data and commercial landings data. ICES assessment working groups assess the status of the main commercial stocks in the North Sea (e.g., cod *Gadus morhua*, haddock *Melanogrammus aeglefinus*, whiting *Merlangius merlangus*, plaice *Pleuronectes platessa* and sole *Solea solea*), although landings data will be available for species that are not subject to stock assessments, and could be examined.

Groundfish surveys provide spatially comprehensive, fishery-independent data for large areas of the North Sea. The main survey data sets available are:

- (a) North Sea quarter 1 IBTS groundfish survey (GOV trawl) which is undertaken by Scotland, Norway, Sweden, Denmark, Germany Netherlands and France
- (b) North Sea quarter 3 IBTS groundfish survey (GOV trawl) by England, Scotland, Norway, Sweden, Denmark and Germany
- (c) North Sea quarter 3 BTS beam trawl survey (4–8m beam trawls) by the Netherlands, England, Germany and Belgium

Other relevant programmes include the Demersal Fish Surveys in coastal waters directed at juvenile flatfish (by England, Belgium, Netherlands and Germany), the Norwegian *Pandalus* survey in the Norwegian Deeps and Skagerrak (150–600m) and various other coastal surveys.

Another appropriate data set would be the epibenthic monitoring project MAFCONS that uses 2m beam trawls during groundfish surveys in the North Sea. This data set could provide better estimates of the abundance of small-bodied demersal species (e.g., gobies and solenette).

A brief review of our current knowledge of the North Sea ichthyofauna, fish distribution, fish communities and biological interactions are given below. Examples of some of the potential outputs that may be generated by analyses of survey data are illustrated by a case study (based on the quarter 3 IBTS data set) in Section 7.3.

7.2.2 North Sea ichthyofauna

The fish fauna of the North Sea has been studied for over 100 years, ranging from the early account by Garstang (1905), to more contemporary studies by Yang (1982a–c), Daan *et al.* (1990), Knijn *et al.* (1993), Greenstreet and Hall (1996), Rice and Gislason (1996), Greenstreet *et al.* (1999), Jennings *et al.* (1999, 2001, 2002) and Callaway *et al.* (2002).

Over 230 fish species have been recorded from the North Sea (Yang 1982a,b; Fricke, 1999; Froese and Pauly, 2003; Table 7.2.1), with FishBase listing 182 fish species (Froese and Pauly, 2003). Yang (1982a) recorded 224 species, which were classified as either dominant (18 species), abundant (8 species), moderately abundant (16 species), frequent (31 species), rare (92 species) or vagrants (59 species). From a biogeographical standpoint, the fish fauna of the North Sea includes northern boreal species (66 species), southern Lusitanian species (110 species) and Atlantic species (48 species) (Yang 1982b).

The North Sea is influenced by three main types of water: North Atlantic oceanic water entering between the Shetland Islands and Norway, North Atlantic waters entering through the English Channel and low salinity water from the Skagerrak (Hill and Dickson 1978). That the hydrography of the North Sea affects the distribution and abundance of marine organisms is well documented, for example increased inflows of oceanic water into the North Sea affect the planktonic community (Edwards *et al.* 1999). Additionally, catches of horse mackerel *Trachurus trachurus* in the northern North Sea during the summer are correlated with the influx of Atlantic water into the North Sea during the preceding winter (Iversen *et al.* 2002). Blue-mouth redfish *Helicolenus dactylopterus* became more abundant in the northern North Sea during 1991 and, in subsequent years, became more widespread in the North Sea (Heessen *et al.* 1996). The reasons for the influx of the 1990–year class of blue-mouth are not known, but may be related to changes in hydrographical factors.

There have been major temporal changes in the ichthyofauna of the North Sea, and these have been discussed at various ICES Symposia (e.g., Hempel 1978). Notable events include the disappearance of bluefin tuna *Thunnus thynnus* from the North Sea, with low catch rates since 1962 (Tiews 1978) and the disappearance of common skate in the southern and central North Sea and associated changes in the elasmobranch assemblage (Walker and Heessen 1996; Walker and Hislop, 1998). The gadoid outburst (from 1962 to the early 1970's) has also attracted much attention, as this period included high recruitment episodes for many gadoids, including cod, haddock and whiting (Cushing 1980, 1984; Hislop 1996). Although the exact causes for this are not known, several hypotheses were suggested, including competitive release (i.e., more food was available), decreased predation on their eggs/larvae following the decline in stocks of pelagic fish (e.g., herring and mackerel), changes in zooplankton dynamics and climatic factors (Cushing 1980, 1984). In the southern North Sea, Attrill and Power (2002) reported that southerly species (e.g., anchovy, wrasse and lesser weever) were more abundant in the Thames Estuary after warmer winters. There are several more recent studies that have used survey data and other information sources to examine long-term changes in fish populations in the North Sea (Tiews, 1990; Greenstreet and Hall 1996; Heessen 1996; Heessen and Daan 1996; Rijnsdorp *et al.* 1996; Rogers and Millner, 1996; Vooy and Meer, 1998; Rogers and Ellis 2000).

Commercial landings of fishes from the North Sea were relatively stable between 1909 and 1961, after which landings increased rapidly (Holden 1978), with landings particularly stable for demersal species and more variable for pelagic species. Species that declined over this period included wolf-fish *Anarhicas lupus*, conger eel, halibut *Hippoglossus hippoglossus*, smelt *Osmerus eperlanus*, and skates and rays. Factors that can lead to changes in landings include changes in (a) environmental conditions (including natural variability) (b) fishing patterns, (c) fishing impacts, and (d) other anthropogenic activities (e.g., pollution). Holden (1978) also suggested that the population centres of several commercial species (haddock, whiting, plaice, sole, turbot *Scophthalmus maximus* and brill *S. rhombus*) had moved northwards. Richards *et al.* (1978) examined trends in research vessel catches (1922–1971) and reported general post-war increases in catch rates of dab, lemon sole and long-rough dab. Postuma (1978) examined landings of southern fish species in the North Sea, specifically pilchard *Sardina pilchardus*, sea-bream *Pagellus centrodontus* (= *P. bogaraveo*) and horse mackerel. Landings of pilchard, although generally small, had peaks (>1000 tonnes) in 1937, 1951, 1958–60, and were mostly landed from the southern North Sea. Sea-bream, in contrast, were most abundant in the northern North Sea, with peak landings in 1967–69.

7.2.3 Distribution of fish species

Although more than 230 species of fish are known from the North Sea, the majority of these species are not encountered in groundfish surveys. This is due to a variety of factors including the sampling grid and gear, and that many of the species are vagrants that will only occur occasionally.

In 1993 an 'Atlas of North Sea Fishes' was published as an ICES Cooperative Research Report (Knijn *et al.* 1993). This atlas was based on data for 98 species (or higher taxa) that were caught during a number of bottom trawl surveys in the summer and winter of the years 1985 to 1987. The 1993 atlas has been out of print for several years and there is a need to prepare an update using more recent data.

The main aim of the 1993 atlas was to give an overview of the data available from trawl surveys and at the same time to fill an important gap in our knowledge of the spatial distribution of North Sea fish species. Prior to the publication of the atlas, information on the spatial distribution of non-commercial species was virtually non-existent, and information on commercial species could only be found in technical reports, which were not readily accessible to non-specialists.

Current initiatives


The 1993 atlas described the situation that prevailed more than 15 years ago. Since that time, a significant amount of new data has become available. In particular, standardised quarterly IBTS Surveys of the North Sea took place during the years 1991–1996. The data from these surveys make it possible to describe and compare seasonal distributions, and to determine whether there have been significant changes in the distribution and abundance of species. At the same time, a complete set of data of the February IBTS since 1965 has become available, which will allow for the analysis of catch trends during winter for a period of more than 30 years. The data are stored in the ICES IBTS Database and will be transferred to the ICES DATRAS Database.

In January 2004 an EU research proposal was submitted by RIVO (Netherlands), CEFAS (England) and ICES to update the 1993 atlas. Due to the very limited budget that was indicated in the call for proposals, it was decided to set up the project in two parts. The 2004 proposal represents phase 1 and has the following objectives:

- revise and update the existing Atlas of North Sea Fishes focusing on a small number of fish species as a pilot project, and
- publish the revised Atlas online.

Phase 2 will only be possible when further funding is secured. It will be a larger project including more partners and its objectives would be to:

- widen the species coverage of the interactive North Sea Fish Atlas to include all species caught in North Sea surveys;
- widen the area coverage of the interactive North Sea Fish Atlas to cover the NE Atlantic; and
- produce a printed "NE Atlantic Fish Atlas"

Similar to the 1993 edition, the online atlas will provide general information on the North Sea ecosystem, information on research vessel surveys, and species-specific information. This will include maps showing (seasonal) distributions, changes in abundance over the past decades, length compositions, information on growth, age, life history, population and exploitation, etc. and provide appropriate references to the scientific literature. Species included in the first phase will cover examples of ecologically different groups, and also represent examples of data-rich and data-poor species. 

The main source of data will be the IBTS survey. Seasonal changes in distribution will be described on the basis of quarterly surveys from the years 1991–1996. Additional information will be derived from other sources available in ICES, such as a number of ICES coordinated surveys targeting flatfish.

When further funding is secured for a second phase, all fish species encountered in the surveys will be included. Also the area covered will be expanded to include the whole NE Atlantic from the southern Iberian Peninsula to Iceland and Norway, this covering the majority of the OSPAR area, and extending where possible into the Baltic and Mediterranean Seas. In addition, the Atlas will be transferred to a server at ICES HQ in Copenhagen with a direct link to the DATRAS database.

It should be mentioned that the ICES atlas-project will mainly rely on financial funding coming available from the European Commission.

In the short term, and for the REGNS (Regional Ecosystem Study Group for the North Sea) project, it would not be practical to provide distribution plots for every species that has been reported from the North Sea. Figures illustrating the distribution and relative abundance of the major fish species (e.g., cod, haddock, grey gurnard *Eutrigla gurnardus*) could be provided for the primary survey gears used (beam trawl and GOV trawl), and plots of the distribution (presence/absence) could also be provided for those species that are regularly encountered in surveys, albeit not effectively (e.g., rocklings). It is suggested that ICES Working Group on Elasmobranch Fishes (WGEF) provides comparable figures for elasmobranch fishes.

For those species where the catch rates from groundfish surveys is appropriate, analyses examining temporal changes in mean latitude and longitude, and spatial-temporal changes in relative abundance could be provided.

Several species of marine fish have either a northern or southern boundary latitude in the North Sea. For such species, if data on relative abundance by latitude/longitude are insufficiently robust for analysis, analyses of maximum/minimum boundary latitude could be provided. Northerly species with a southern boundary in the North Sea include wolf-fish *Anarhichas lupus*, Norway pout *Trisopterus esmarki*, long-rough dab *Hippoglossoides platessoides* and certain species of eelpout. Southerly species with a northern boundary in the North Sea may include red mullet *Mullus surmuletus*, anchovy *Engraulis encrasicolus* and lesser weever *Echiichthys vipera*.

7.2.4 Fish assemblages in the North Sea

Regarding fish communities, our perception of fish *communities* from trawl surveys is heavily biased by the gear used. Hence, the use of groundfish surveys is more appropriate for examining species composition and diversity of fish *assemblages*. Spatial and temporal patterns in the species diversity and structure of the major assemblages could be provided. Other aspects of the community (e.g., size spectra) could also be examined (see Section 2).

The fish assemblages in the northern North Sea (ICES Division IV a) and northern parts of the Central North Sea (ICES Division IV b) are very different to those further south (Callaway *et al.*, 2002), with this latitudinal gradient mirrored with corresponding gradients in depth and water temperature. The dominant fish species in trawl catches include whiting *Merlangius merlangus* and haddock *Melanogrammus aeglefinus*, and pelagic species including mackerel *Scomber scombrus* and horse mackerel *Trachurus trachurus* in the summer. In shallower waters (50–100m depth), populations are dominated by haddock, whiting, herring *Clupea harengus*, dab *Limanda limanda* and plaice *Pleuronectes platessa*, while at greater depth (100–200m), Norway pout *Trisopterus esmarki* dominate (Callaway *et al.*, 2002). The northern North Sea also contains a number of boreoarctic species that are rarely found further south (e.g., Vahl's eelpout *Lycodes vahlii* and Esmark's eelpout *L. esmarkii*). Further north, and in the Norwegian Deep, various species of deep-water fish (e.g., macrourids) are more abundant.

The southern North Sea (ICES Division IVc), is generally shallow (<50m deep) and the dominant fish species are those that are more characteristic of inshore waters. Plaice *Pleuronectes platessa*, sole *Solea solea*, dab *Limanda limanda* and whiting *Merlangius merlangus* are some of the dominant commercial species, and non-commercial species such as lesser weever *Echiichthys vipera*, grey gurnard *Eutrigla gurnardus* and solenette *Buglossidium luteum* are also an important component of the fish assemblage (Callaway *et al.*, 2002). Sandeels (Ammodytidae) and sand gobies

(*Pomatoschistus* spp.), which are poorly sampled by research trawls, are also abundant and are important prey species for many species of demersal fish. Other recent studies examining the fish assemblages of the North Sea include Rogers *et al.* (1998, 1999).

Other important fish communities/assemblages that are sampled by some national and international surveys include estuarine and inshore fish communities (e.g., Elliott and Hemingway, 2002). Those species of fish that occur in estuarine environments in the North Sea region, but are not generally encountered in marine environments, are listed in Table 7.2.2. Examining the status of fish communities in transitional waters would provide valuable information to REGNS and may also help address the Water Framework Directive.

7.2.5 Species of conservation importance

OSPAR and other fora (e.g., Habitat's Directive), have listed several species of fishes that occur in the North Sea as "threatened and declining", including elasmobranchs (e.g., common skate *Dipturus batis* and spotted ray *Raja montagui*), diadromous fishes (e.g., sea lamprey *Petromyzon marinus*, Allis shad *Alosa alosa*, Surgeon and houting *Coregonus lavaretus oxyrinchus*) and commercial species (e.g., cod *Gadus morhua*). The status of these species in the North Sea may serve as another useful indicator for the health of the North Sea ecosystem.

Common skate occurs primarily off the Shetland Isles (Ellis *et al.*, in press), although it was formerly much more widespread. Analyses of elasmobranch fishes in the North Sea would be an important element of monitoring the status of North Sea fish communities, and should be undertaken by WGEF.

Similarly, diadromous fish species should be examined by the ICES Study Group on the Status of Diadromous Fish Species (SGSDFS). Although there are currently no internationally-coordinated surveys of estuarine fishes, the Water Framework Directive is an important policy driver, and most nations bordering the North Sea will have monitoring programmes in the major estuaries and/or coastal waters. The use of these data sets to examine the status and trends of diadromous fish species, and estuarine fish communities in general, should be investigated by REGNS, as such data will provide information on the status of threatened species and may also serve as an index of the health and quality of the estuarine and transitional waters feeding into the North Sea.

Other species that occur in the northern North Sea, and that may become of conservation interest in the future, include wolf-fish *Anarhichas lupus*. This large-bodied demersal species has a comparatively low fecundity, and similar taxa in Canadian Atlantic waters have declined dramatically (O'Dea and Haedrich, 2003). Examinations of available data (survey data and landings data) for such species could be provided.

7.2.6 Biological interactions

Two major international studies of fish predator-prey interactions have been carried out in the North Sea, the 1981 and 1991 "year of the stomach" projects, co-ordinated by ICES (Daan 1989; Hislop 1997). The main purpose of these projects was to provide the information necessary to parameterise the Multi-Species Virtual Population Analysis models (MSVPA), consequently sampling effort was principally directed at the five main commercial piscivorous fish predators, cod, haddock, whiting, saithe and mackerel. In the later study some data were collected covering a few additional abundant non-commercial predators, such as grey gurnard. Few studies have utilised these "year of the stomach" data within a broader ecosystem context. In a recent revision of earlier North Sea food web studies (Steele 1974; Jones 1982; Jones 1984), Greenstreet *et al.* (1997) used "year of the stomach" data to estimate seasonal variation in the consumption of twelve different major prey types by demersal and pelagic piscivorous fish in the North Sea. This study was part of the process involved in parameterising the European Regional Seas Ecosystem Model (ERSEM) (Bryant *et al.* 1995). Frid *et al.* (1999) utilised the same data to demonstrate that predation levels on benthic invertebrate prey had declined over the period 1981 to 1991.

More recent evaluation of the predator-prey interactions involving the major gadoid predators in the North Sea is reported more fully in section 4 of this report. A new project BECAUSE (Critical Interactions Between Species and their Implications for Precautionary Fisheries Management in a variable Environment – a modelling approach), which will examine key predator prey interactions in the North Sea, and other marine regions, has recently started and should be reporting results by 2006/2007. This project is also briefly described in Section 4.

Contemporary data regarding the feeding ecology of many species in the North Sea are lacking, and this is particularly the case with respect to non-commercial species. Few, if any, studies at a North Sea spatial scale, or anywhere approaching a middle to long-term time scale, have been carried out for many of the most abundant species in the North

Sea, such as common dab, long rough dab, and Norway pout. The working group is, however aware of some work currently in progress that may address at least some of these shortcomings.

The role of competitive interactions within fish communities in the North Sea is not clear, and this remains a major line of enquiry. The MAFCONS (Managing fisheries to conserve groundfish and benthic invertebrate species diversity) project will test hypotheses derived from Huston's (1994) dynamic equilibrium model, which is itself firmly founded in competition theory. The food web studies cited above suggest that energy supply to the benthic part of the North Sea food web may be in short supply, whilst the energy requirement for the pelagic food chains are adequately supplied. If strong competitive interactions are to be found anywhere in the North Sea fish community, these studies suggest that they will be observed in the demersal fish assemblage.

7.3 Case study: North Sea quarter 3 groundfish survey (FRS)

7.3.1 Introduction

The term of reference given to WGFE specifically asked the working group to “start preparations to summarise status and changes in fish species distribution and fish community composition and interactions in the North Sea for the period 2000 to 2004”. WGFE felt that this time frame was too short for any meaningful changes to be observed in many fish stocks, and that for many species of fish temporal analyses covering such a narrow time period could result in misleading trends for the population as a whole. By “status”, WGFE assumed that some sort of indication of current population abundance and demography was required. Stock assessments of the commercial species are carried out on an annual basis and these provide estimates of stock size in each year based on landings data. WGFE took the view that the working groups involved in these stock assessments were the more appropriate fora to summarise this information for input to REGNS. However, the number of species assessed in this way is limited. For the majority of species recorded in the North Sea, survey data provides the only regional indicator of trends in abundance. WGFE therefore concentrated on the analysis of groundfish survey data, e.g., the ICES co-ordinated quarter 3 (Q3) international bottom trawl survey (IBTS).

For the purposes of this illustrative example, the analyses presented included only the major demersal fish in the North Sea. The Q3 IBTS is a demersal otter trawl survey and whilst the gear does sample pelagic species such as mackerel *Scomber scombrus* and herring *Clupea harengus*, the catchability of these species in the gear differs from most demersal species. The stochastic nature of the catches, resulting from by the shoaling habits of pelagic species, makes interpretation of the data problematic. WGFE proposes to address these issues in future meetings, should it be required. Five of species analysed here were commercial species. Trends in the abundance of these species could be compared to recent stock assessment output to see how well indices of abundance derived from survey data performed relative to other abundance estimates. Such comparisons need to incorporate issues such as the relative catchability of different species in particular trawl gears (see Section 6).

The requirement to summarise “changes in fish species distribution” was clear, and preliminary analyses examining the distributions of nine of the most abundant demersal species in the North Sea were undertaken. The four-year period that the working group was set seemed too short to really examine temporal trends. Consequently we present data covering the period 1998 to 2003. During this period all countries participating in the Q3 IBTS used the GOV trawl. Prior to this date, some participants used other trawl gears. Data for 2004 are not yet available. The distributions of the different species were compared and changes over time examined, although it is reiterated that the time period used may not be sufficient to accurately identify trends in abundance for some species.

The ToR also required WGFE to summarise changes in “fish community composition”. This section of the ToR was less clearly defined. WGFE therefore adopted the following approach. Catches from trawl surveys do not sample all fish species with comparable effectiveness, and not all habitats are sampled, hence, the use of survey data is better used to examine assemblages, i.e., groups of fish species with similar spatial patterns in distribution and relative abundance. Cluster analysis of the species abundance data can identify groups of ICES statistical rectangles where patterns of species relative abundance are similar, and those rectangles where catches are different. Thus in each year, all statistical rectangles in the IBTS area could be assigned to a particular “fish assemblage”. Variation in the boundaries of these “assemblages” from year to year was examined.

“Fish Communities” is one of the 10 Ecological Quality issues for which Ecological Quality Objectives will be set. The current element of Ecological Quality identified in the Bergen Declaration for fish communities concerns “changes in the proportion of large fish and hence the average weight and average maximum length of the fish community”. Trends in the mean weight of fish in each of the fish communities identified by the cluster analysis are therefore presented. These data can provide “baseline” information against which the effectiveness of management action to “restore” degraded communities might be judged. Maps of the spatial variation of this parameter are also provided. Other

characteristics of fish communities may also be of relevance given other “policy drivers” underpinning the development of the ecosystem approach to management. For example, commitments to the Convention on Biological Diversity, Agenda 21 and Annex V of OSPAR imply that the conservation and restoration of species diversity in marine communities is also required. These datasets can also be used to provide data for similar analyses examining several indicators of species diversity.

7.3.2 Methods

ICES International Bottom Trawl Survey (IBTS) data for quarter 3 (Q3) for the period 1998 to 2003 were collated into two databases, Haul Summary Information and Species Abundance at length data. Only hauls of exactly 30 minutes duration were analysed to keep the effects of effort variation to a minimum.

To calculate fish density, estimates of the area sampled were required. Gear geometry data i.e., door-spread, wing-spread, headline height (Figure 7.3.2.1), were obtained using SCANMAR© recording units. Two density estimates were determined. Firstly the area swept by the whole gear i.e., between the otter boards, is given by:

- Gear Swept Area (GSA) $m^2 = \text{Mean Door Spread (m)} \times \text{Distance Towed (m)}$

Secondly the area swept by the net, i.e., between the wing ends of the trawl, is given by:

- Net Swept Area (NSA) $m^2 = \text{Mean Wing Spread (m)} \times \text{Distance Towed (m)}$

Data on door and wing-spread were not available for a substantial number of the hauls included in the analysis. Scottish SCANMAR© data for the years 1998, 2001, 2002 and 2003 were analysed to determine the relationships between the two parameters and depth. The relationships in all four years were similar so the regression analysis was performed on the combined data (Figure 7.3.2.2). Where no depth data were available for a trawl sample, the average depth recorded on other occasions that the station was fished was applied. The distance towed for each haul is not a mandatory value in the ICES database. Where this value was missing the average distance towed for all trawls in that particular year was used. Once all missing values had been added, the two area swept parameters could be determined using the following equations:

$$\text{Mean Door Spread} = 33.251 \times \log \text{depth} + 15.744$$

$$\text{Mean Wing Spread} = 6.8515 \times \log \text{depth} + 5.8931$$

Data in the ICES database are stored as the number of fish caught in each length class raised to the hours fishing effort. In order to estimate densities in the original 30min tows all numbers at length were divided by 2. Fish density was calculated using the following formula:

$$\text{Fish density (nos. } m^{-2}) = \text{total number of fish per } \frac{1}{2} \text{ hour tow} / \text{swept area measure (m}^2)$$

In most years some statistical rectangles were fished more than once. In these cases the total number of fish caught in all samples in the rectangle was divided by the total area swept by all trawls in the rectangle combined to give the fish density estimate for that particular statistical rectangle.

To estimate biomass density, weight-at-length data are necessary. Since such data are not available in the ICES database, weight-at-length relationships maintained by FRS Marine Laboratory, Scotland, were used. These relationships were used to convert numbers at length in the database to weight at length. Biomass density was then determined as:

$$\text{Fish biomass density (Kg. } m^{-2}) = \text{Total weight of fish per } \frac{1}{2} \text{ hour tow} / \text{swept area measure (m}^2)$$

As we do not know the catchability of the various species, which is also affected by many factors (see section 6), it is not possible at the current time to raise these biomass densities to absolute values. Hence catch data have been transformed to area-weighted catches.

In order to estimate the area-weighted catch of fish present in each entire statistical rectangle, these density estimates need to be multiplied by the “sea area” of the rectangle. As you move further north, the width of each statistical rectangle decreases due to the curvature of the earth. Taking this into account the area of each rectangle was determined and then multiplied by the proportion that was sea to give the “sea area” in each ICES rectangle. The area-weighted

catch of each species present in each ICES statistical rectangle could then be estimated by multiplying species density by the sea area of each rectangle. Total species area-weighted catch estimates for each year were determined by summing the rectangle biomass estimates over all rectangles.

In each of the six years there were at least two statistical rectangles where no fishing took place. For these missing rectangles an area-weighted catch was interpolated based on the mean of the area-weighted catch estimates in the surrounding statistical rectangles.

Figure 7.3.2.3 shows the full extent of ICES area IV. The rectangles shaded light blue were included in the IBTS in at least one of the six years and were therefore included in the analysis. Areas shown in white were not included in the analysis so the biomass could not be estimated. The biomass estimates determined from the IBTS for the whole North Sea are for an area slightly smaller than the full ICES area IV. In order to compare these IBTS based area-weighted catch estimates with those derived for the full ICES area IV by the ICES Working Group on the Assessment of Demersal Stocks in the North Sea and Skagerrak (WGNSSK), the IBTS area-weighted catch estimates were adjusted by a raising factor determined by the ratio:

$$\text{Raising Factor} = \text{ICES Area IV area (m}^2\text{)} / \text{Area covered by the IBTS survey (m}^2\text{)}$$

To take account of the fact that fish were not evenly distributed across the North Sea, raising factors were determined for the five separate zones indicated in Figure 7.3.2.3 (Table 7.3.2.1).

Figure 7.3.2.4 shows the total number of hauls in each statistical rectangle within the study area in each year and Figure 7.3.2.5 shows the total number of times each statistical rectangle was fished over the six-year period 1998 to 2003. Only fish that were considered to be demersal were included in these analyses. The species caught in the IBTS that were considered to be demersal are given in Table 7.2.1.

7.3.3 Results

The analyses presented here use density estimates derived using the area swept by the net. Densities of species that may have been herded into the path of the net to a significant extent by the action of the otter boards and sweeps may be over-estimated relative to species not susceptible to such herding effects. Data are presented for nine species: haddock *Melanogrammus aeglefinus*, whiting *Merlangius merlangus*, cod *Gadus morhua*, Norway pout *Trisopterus esmarki*, grey gurnard *Eutrigla gurnardus*, plaice *Pleuronectes platessa*, lemon sole *Microstomus kitt*, common dab *Limanda limanda*, and long rough dab *Hippoglossoides platessoides*. These nine species account for between 91% and 98% of the individual fish sampled from the demersal component of the fish assemblage in each year by the IBTS (Table 7.3.3.1).

Distribution

Figures 7.3.3.1 to 7.3.3.9 illustrate variation in the catch rates of each of the nine species in each year in each ICES rectangle covered by the IBTS. Haddock was mainly confined to the north-western North Sea in all six years (Figure 7.3.3.1). Whiting was more ubiquitous, but the largest biomasses tended to occur off the east coast of Britain and in the southern North Sea (Figure 7.3.3.2). Cod was also found over most of the North Sea, albeit in low numbers, but the largest biomass density occurred in the north-eastern North Sea (Figure 7.3.3.3). Norway pout was mainly limited to the northern half of the North Sea in most years, although there was some indication of a southerly expansion of the distribution in 2000 (Figure 7.3.3.4). Figure 7.3.3.5 shows the distribution of Grey gurnard relative abundance over the whole North Sea. Grey gurnard was found over much the IBTS area, but was most abundant across the centre of the North Sea. Catch rates were low both north of 57.5°N and south of 53.5°N (Figure 7.3.3.5). Plaice was found over much of the area, but was much less abundant in the north-eastern North Sea (Figure 7.3.3.6). The largest catch rates of lemon sole were located in the northern North Sea, just to the south of the Shetland Isles. Lemon sole biomass in the south-eastern North Sea was low (Figure 7.3.3.7). Common dab was mainly confined to the southern half of the IBTS area, with biomass highest in the south-east and central North Sea, particularly off the coasts of Denmark and Holland (Figure 7.3.3.8). Long rough dab was most abundant in the central North Sea. Whilst their distribution extended into the northern North Sea, almost no long rough dab were sampled south of 53.5°N (Figure 7.3.3.9). The actual values of area-weighted total catch for each species are given in Figures 7.3.3.10 to 7.3.3.18.

Centres of geographic distribution

Geographic centroids were determined for each species in each year. Centroids were defined by the biomass-weighted mean latitude and longitude to establish the centre points, along with the northerly, southerly, easterly and westerly

quartiles for the population as a whole. In those cases where fishes in the North Sea are distributed around a single, central area of peak abundance, the geographical centroids will mirror the areas of peak distribution. However, for those species with skewed or patchy distributions, the geographical centroids represent the mean latitude/longitude for the North Sea population as a whole, and do not reflect their true distribution. Nevertheless this is a method that may have utility in examining subtle shifts in species distribution.

These centroids are illustrated in Figures 7.3.3.19 to 7.3.3.27. These centroids allow changes in distribution to be assessed. Thus the haddock distribution appears to have shifted slightly northwards in 1999, the year of the large recruitment cohort (Figure 7.3.3.19). The whiting distribution may have shifted in a north-easterly direction in 1999, and in a south-easterly direction in 2001 (Figure 7.3.3.20). For this species the centroids are skewed by their predominance in the western and south-eastern North Sea. The centroids for cod were wide-ranging and heavily skewed by the patchy distribution of the species at the present time (Figure 7.3.3.21). The distribution of Norway pout appears to have shifted slightly to the north-east in 2002 and 2003 (Figure 7.3.3.22). There appears to have been little movement in the central location of the grey gurnard distribution (Figure 7.3.3.23). The geographical centroid for plaice has shifted westwards (Figure 7.3.3.24). The distribution of lemon sole may have shifted further to the north-east in 2000 and 2001 (Figure 7.3.3.25). There has been little movement in the common dab centroid location or area over the period 1998 to 2003 (Figure 7.3.3.26). The centroids for long rough dab were generally in the central North Sea (Figure 7.3.3.27).

The centroid boundaries for the four gadoid species in all years are shown in Figure 7.3.3.28. This illustrates clear differences in the averaged latitudinal and longitudinal locations of these four species. Norway pout, haddock and whiting have distinctly westerly distributions, with Norway pout occupying more northerly regions, haddock occupying an intermediate zone, and whiting more abundant in the central western North Sea. Cod have a more diffuse and patchy distribution than the three other abundant gadoids, occupying the northeasterly North Sea, with other patches off northeast England and, in some years, in the southern North Sea. Figure 7.3.3.29 shows the centroid boundaries for the four flatfish species and grey gurnard. In contrast to the gadoids, clear overlap in the distributions of some of these species is indicated. Plaice, common dab and grey gurnard all primarily occupy the central eastern North Sea, with the gurnard distribution firmly centred on the western half of the two flatfish distributions, which both overlap almost completely. The distributions of long rough dab and lemon sole overlap little with each other, or with the other two flatfish. The long rough dab centroids have a more northerly location, while lemon sole are mainly centred further to the north-west.

Area weighted total catch

Summing the individual rectangle density estimates across all ICES rectangles in the IBTS survey area (including interpolated values) provided IBTS derived estimates of the area-weighted total catch of each species in the area in each year. Table 7.3.3.2 gives the results of this summing procedure for each of the five zones of the North Sea (see Figure 7.3.2.3) and also provides the raising factors for each zone to raise these totals to the whole of ICES area IV. Table 7.3.3.2 then gives the results following the application of these raising factors for each zone, and gives the area totals (IBTS area and ICES area IV) after summing across the five zones. Annual variation in the ICES area IV totals for each of the nine species is illustrated in Figure 7.3.3.30.

The area weighted total catch of haddock increased to a peak of 252,000t in 2000, then subsequently declined. Whiting increased to a peak of 177,000t in 2001 and has then decreased in the following two years. Cod decreased from 19,600t in 1998 to 10,000t in 2000, and rose slightly again in 2001 then decreased sharply to 5,600t in 2003. Norway pout rose to a peak of 126,000t in 2000 and has decreased sharply since.

The area weighted total catch for grey gurnard has remained relatively constant over the six-year period, fluctuating around 20,000t. Plaice increased to a peak of 7,000t in 1999, then decreased to a low of under 4000t in 2000, and has subsequently increased. Lemon sole has remained relatively constant at around 5,500t. Common dab was relatively stable at 35,000t from 1998 to 2000, then increased rapidly to just under 60,000t in 2001, and has remained relatively stable since then. Long rough dab has remained relatively constant over the six-year period at about 6,000t.

With some exceptions, the trends in abundance indicated by the IBTS follow closely the ICES stock assessment estimates. The comparison in the ICES area IV biomass estimates for cod, haddock, whiting, Norway pout and plaice in each year derived from the IBTS and the equivalent annual stock assessments made by the WGNSSK (Working Group on Assessment of Demersal Stocks in the North Sea and Skagerrak) are shown in Figure 7.3.3.31. The peak haddock biomass indicated by the stock assessment is picked up in the IBTS signal, but the amplitudes differ. This difference may be caused by over estimation of the large 1999 recruit cohort strength by the assessment process, and the difference may decline with future iterations of the VPA analysis. Otherwise the haddock biomass estimates derived from the VPA are in close agreement with the estimates derived solely from the IBTS. The two whiting abundance trends follow similar trajectories, but the IBTS estimates are a factor of two to three times higher, suggesting a catchability in the

GOV trawl of >1 . This may be due to the herding effect of the otter boards on whiting in the path of the gear, i.e., a much larger area is being sampled for whiting than is allowed for by the Net Swept Area. The distance between the otter boards is approximately 3.5 times greater than the distance between the trawl wing ends. If Gear Swept Area is used instead to estimate whiting densities, then the IBTS biomass estimates are approximately half the value of the assessment stock biomass estimates. For all other species, the IBTS derived biomass estimates are lower than the equivalent stock assessment estimates suggesting catchabilities of <1 in the IBTS. In the case of plaice, the assessment biomass estimates exceed the IBTS derived estimates by an order of magnitude. This may reflect the low catchability of the GOV trawl for certain flatfish species (see Section 6). Norway pout stock assessment estimates are approximately 6 times higher than the equivalent IBTS biomass estimates. The larger mesh size used by the IBTS compared with the industrial fishery may help to explain part of this difference.

Fish Assemblages

In each year, the total number of individual fish in each species present in each statistical rectangle was estimated. Cluster analysis was performed on these data to group together rectangles with similar species composition, and distinguish between groups of rectangles with different species relative abundance composition. An example of the dendrogram produced by this analysis is provided in Figure 7.3.3.32, in this case for the year 1998. Similar dendrograms were obtained in all other years, but are not illustrated here. For each year the main clusters, potentially different fish assemblages were defined at a Bray-Curtis similarity of about 35%. This level of similarity left a single large single fish assemblage cluster located across most of the central and northern North Sea. This cluster broke down into two consistent sub-clusters at a similarity of around 40% (Figure 7.3.3.33). The clustering of the species abundance data was reasonably consistent from year to year, strongly suggesting the presence of distinct fish assemblages. These assemblages tended to exist as bands straddling the North Sea more or less with a south-west to north-east orientation. The locations of the boundaries between assemblages were reasonably stable in time. The data from all years were combined to produce a composite indication of the spatial extent of each fish assemblage cluster across all six years (Figure 7.3.3.34).

Mean weight of fish

In each year the average weight of demersal fish in each rectangle was estimated (Figure 7.3.3.35). The largest fish were found around the 200m depth contour towards the north of the IBTS region and the along the western edge of the Norwegian deeps. Mean weight appeared to be higher in a region extending out in to the central North Sea from the eastern coast of the UK. For each of the five fish assemblages identified in Figure 7.3.3.34, trends in the mean size of demersal fish over the six-year period for which data were analysed are shown in Figure 7.3.3.36. Both positive and negative slopes were detected but none were statistically significant. This is not surprising given the small sample size. These data could provide baseline information from which to monitor the effects of remedial action taken to mitigate long term declines in these community indicators.

7.4 Conclusions

- The analyses presented above were carried out entirely on the Q3 IBTS data-set, and only included demersal fish species. Numerous other data-sets are available, for examples see Sections 6 and 7.2.1. Similar analyses could be carried out on these data-sets for comparative purposes and some earlier data sets could be used to examine seasonal variation.
- These analyses have been carried out entirely for illustrative purposes to indicate the sorts of information that can be obtained from groundfish survey data. No effort has been made to explain any apparent trends, for example apparent between year variation in species distributions. However, clearly gaining some understanding of the underlying mechanisms involved will be of great importance in implementing an ecosystem approach to management and undertaking regional ecosystem assessments.
- Time restrictions prevented the working group from examining other possible community metrics or indicators. Section 2.4.2 examines a suite of twelve community metrics. Each of these potential indicators of fish community status could be investigated in a similar way to the approach adopted here for mean weight of fish.
- WGFE carried out an extensive analysis in the case study presented here. The reason for this was to illustrate to REGNS, as far as possible, the range of potential analyses available. In this way the working group hoped that the feedback from REGNS would be equally extensive, so that future work towards a regional ecosystem assessment for the North Sea might be more tightly focused.
- Although 2000–2004 survey data could be used to highlight the current status of the distribution and relative abundance of a large number of fish species in the North Sea, the assessment of temporal trends should incorporate longer time periods.

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Table 7.2.1 Checklist of marine fish species in the North Sea (Adapted from Yang (1982a), Fricke (1999), FishBase and the IBTS database). The qualitative listings of occurrence(*) follow that of Yang (1982a), with those species listed on the IBTS database and not substantiated by another study indicated as “Status uncertain”). An updated index is required for many species. Those demersal species which are listed in the IBTS database and for which length-weight data are available are indicated in the final column, and taxa which are sometimes recorded at a higher taxonomic level (e.g., Family or Genus) are indicated \$.

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Myxinidae	<i>Myxine glutinosa</i>	Hagfish	Atlantic	Frequent	✓	Y
Petromyzontidae\$	Lampetra fluviatilis	River lamprey	Boreal	Rare	✓	N
Petromyzontidae\$	<i>Petromyzon marinus</i>	Sea lamprey	Boreal	Rare	✓	Y
Chimaeridae	<i>Chimaera monstroza</i>	Rabbit fish	Atlantic	Rare	✓	Y
Hexanchidae	<i>Hexanchus griseus</i>	Six-gilled shark	Atlantic	Vagrant	✓	
Alopiidae	<i>Alopias vulpinus</i>	Thresher shark	Atlantic	Vagrant	✓	
Cetorhinidae	<i>Cetorhinus maximus</i>	Basking shark	Atlantic	Rare	✓	
Lamnidae	<i>Isurus oxyrinchus</i>	Mako shark	Atlantic	Vagrant		
Lamnidae	<i>Lamna nasus</i>	Porbeagle shark	Atlantic	Rare	✓	
Scyliorhinidae	<i>Galeus melastomus</i>	Blackmouthed dogfish	Lusitanian	Frequent	✓	Y
Scyliorhinidae	<i>Scyliorhinus canicula</i>	Lesser-spotted dogfish	Lusitanian	Moderately abundant	✓	Y
Scyliorhinidae	<i>Scyliorhinus stellaris</i>	Nurse hound	Lusitanian	Rare	✓	Y
Triakidae	<i>Galeorhinus galeus</i>	Tope shark	Lusitanian	Frequent	✓	Y
Triakidae	<i>Mustelus asterias</i>	Starry smooth hound	Lusitanian	Frequent	✓	N
Triakidae	<i>Mustelus mustelus</i>	Smooth hound	Lusitanian	Rare	✓	N
Carcharhinidae	<i>Prionace glauca</i>	Blue shark	Atlantic	Vagrant	✓	
Sphyrnidae	<i>Sphyrna zygaena</i>	Common hammerhead	Atlantic	Vagrant		
Dalatiidae	<i>Dalatias licha</i>	Darkie charlie	Atlantic	Rare		
Dalatiidae	<i>Etmopterus spinax</i>	Velvet belly	Atlantic	Rare	✓	N
Dalatiidae	<i>Somniosus microcephalus</i>	Greenland shark	Boreal	Vagrant	✓	
Squalidae	<i>Squalus acanthias</i>	Spurdog	Atlantic	Abundant	✓	Y
Echinorhinidae	<i>Echinorhinus brucus</i>	Bramble shark	Circumglobal	Vagrant	✓	
Squatinae	<i>Squatina squatina</i>	Angel shark	Lusitanian	Vagrant	✓	Y
Torpedinidae	<i>Torpedo mar-morata</i>	Marbled electric ray	Lusitanian	Vagrant	✓	
Torpedinidae	<i>Torpedo nobiliana</i>	Common electric ray	Lusitanian	Vagrant	✓	
Rajidae\$	<i>Amblyraja radiata</i>	Starry ray	Boreal	Dominant	✓	Y
Rajidae\$	<i>Dipturus batis</i>	Common skate	Boreal	Frequent	✓	Y
Rajidae\$	<i>Dipturus nidarosiensis</i>	Black skate	Boreal	Rare		
Rajidae\$	<i>Dipturus oxyrinchus</i>	Long-nose skate	Lusitanian	Rare		Y
Rajidae\$	<i>Leucoraja circularis</i>	Sandy ray	Lusitanian	Moderately abundant		Y
Rajidae\$	<i>Leucoraja fullonica</i>	Shagreen ray	Boreal	Moderately abundant	✓	Y

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Rajidae\$	<i>Leucoraja naevus</i>	Cuckoo ray	Lusitanian	Abundant	✓	Y
Rajidae\$	<i>Raja brachyura</i>	Blonde ray	Lusitanian	Rare		Y
Rajidae\$	<i>Raja clavata</i>	Thornback ray	Lusitanian	Frequent	✓	Y
Rajidae\$	<i>Raja microocellata</i>	Painted ray	Lusitanian	Vagrant	✓	
Rajidae\$	<i>Raja montagui</i>	Spotted ray	Lusitanian	Frequent		Y
Rajidae\$	<i>Raja undulata</i>	Undulate ray	Lusitanian	Vagrant		
Rajidae\$	<i>Rajella fyllae</i>	Round skate	Boreal	Vagrant		
Myliobatidae	<i>Myliobatis aquila</i>	Eagle ray	Lusitanian	Vagrant	✓	
Dasyatidae	<i>Dasyatis pastinaca</i>	Sting ray	Lusitanian	Rare	✓	Y
Acipenseridae	<i>Acipenser sturio</i>	Sturgeon	Lusitanian	Rare	✓	Y
		North American	Atlantic			
Acipenseridae	<i>Acipenser oxyrinchus</i>	sturgeon	North Atlantic,	Vagrant		
Anguillidae\$	<i>Anguilla anguilla</i>	European eel	Baltic Sea	Frequent	✓	N
Muraenidae	<i>Muraena helena</i>	Moray eel	Atlantic	Vagrant		
Congridae	<i>Conger conger</i>	European conger eel	Lusitanian	Rare	✓	Y
Clupeidae	<i>Alosa alosa</i>	Allis shad	Lusitanian	Rare	✓	
Clupeidae	<i>Alosa fallax</i>	Twaite shad	Lusitanian	Rare	✓	
Clupeidae	<i>Clupea harengus</i>	Herring	Boreal	Dominant	✓	
Clupeidae	<i>Sardina pilchardus</i>	Pilchard	Lusitanian	Dominant	✓	
Clupeidae	<i>Sprattus sprattus</i>	Sprat	Lusitanian	Dominant	✓	
Engraulidae	<i>Engraulis encrasicolus</i>	European anchovy	Lusitanian	Rare	✓	
Argentinidae	<i>Argentina silus</i>	Gt silver smelt	Boreal	Abundant	✓	
Argentinidae	<i>Argentina sphyraena</i>	Lsr silver smelt	Lusitanian	Dominant	✓	
Osmeridae	<i>Osmerus eperlanus</i>	Smelt	Boreal	Rare	✓	
Osmeridae	<i>Mallotus villosus</i>	Capelin	Boreal	Vagrant		
Salmonidae	<i>Coregonus oxyrinchus</i>	Houting	Boreal	Rare	✓	
Salmonidae	<i>Salmo salar</i>	Atlantic salmon	Atlantic	Frequent	✓	
Salmonidae	<i>Salmo trutta</i>	Trout	Boreal	Rare	✓	
Salmonidae	<i>Salvelinus alpinus</i>	Charr	Boreal	Rare	✓	
Sternoptychidae	<i>Argyropsalectus olfersi</i>	Hatchet fish	Atlantic	Rare	✓	
Sternoptychidae	<i>Maurolicus muelleri</i>	Pearlside	Atlantic	Rare	✓	
Myctophidae	<i>Myctophum punctatum</i>	Spotted lanternfish	North Atlantic	Vagrant	✓	
Nemichthyidae	<i>Nemichthys scolopaceus</i>	Slender snipe eel	Circumglobal	Vagrant	✓	
Paralepididae	<i>Paralepis coregonoides</i>	Sharpchin barracudina	North Atlantic	Vagrant	✓	
Gadidae	<i>Gadicius argenteus</i>	Silvery pout	Lusitanian	Dominant	✓	

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Gadidae	<i>Gadus morhua</i>	Cod	Boreal	Dominant	✓	Y
Gadidae	<i>Melanogrammus aeglefinus</i>	Haddock	Boreal	Dominant	✓	Y
Gadidae	<i>Merlangius merlangus</i>	Whiting	Lusitanian	Dominant	✓	Y
Gadidae	<i>Micromesistius poutassou</i>	Blue whiting	Atlantic	Moderately abundant	✓	
Gadidae	<i>Pollachius pollachius</i>	Pollack	Lusitanian	Moderately abundant	✓	Y
Gadidae	<i>Pollachius virens</i>	Saithe	Boreal	Dominant	✓	Y
Gadidae	<i>Raniceps raninus</i>	Tadpole fish	Boreal	Rare	✓	Y
Gadidae	<i>Trisopterus esmarkii</i>	Norway pout	Boreal	Dominant	✓	Y
Gadidae	<i>Trisopterus luscus</i>	Bib	Lusitanian	Rare	✓	Y
Gadidae	<i>Trisopterus minutus</i>	Poor cod	Lusitanian	Moderately abundant	✓	Y
Phycidae	<i>Phycis blennoides</i>	Greater forkbeard	Lusitanian	Rare	✓	Y
Lotidae\$	<i>Antonogadus macrophthalmus</i>	Big-eyed rockling	Lusitanian	Status uncertain		Y
Lotidae\$	<i>Brosme brosme</i>	Tusk	Boreal	Moderately abundant	✓	Y
Lotidae\$	<i>Ciliata mustela</i>	5-bearded rockling	Boreal	Frequent	✓	Y
Lotidae\$	<i>Ciliata septentrionalis</i>	Northern rockling	Boreal	Rare	✓	
Lotidae\$	<i>Enchelyopus cimbrius</i>	4-bearded rockling	Boreal	Frequent	✓	Y
Lotidae\$	<i>Gaidropsarus mediterraneus</i>	Shore rockling	Lusitanian	Uncertain	✓	Y
Lotidae\$	<i>Gaidropsarus vulgaris</i>	3-bearded rockling	Lusitanian	Rare	✓	Y
Lotidae\$	<i>Molva dypterygia</i>	Blue ling	Boreal	Rare	✓	Y
Lotidae\$	<i>Molva molva</i>	Common ling	Boreal	Moderately abundant	✓	Y
Merlucciidae	<i>Merluccius merluccius</i>	European hake	Lusitanian	Frequent	✓	Y
Macrouridae\$	<i>Caelorinchus caelorhincus</i>	Hollow nosed rattail	Atlantic	Rare		
Macrouridae\$	<i>Coryphaenoides rupestris</i>	Roundhead rat-tail	Atlantic	Rare	✓	
Macrouridae\$	<i>Macrourus berglax</i>	Rough rat-tail	Boreal	Rare		
Macrouridae\$	<i>Malacocephalus laevis</i>	Softhead rattail		Status uncertain		Y
Carapidae	<i>Echiodon drummondii</i>	Pearlfish	Lusitanian	Rare	✓	Y
Lophiidae\$	<i>Lophius budegassa</i>	Black-bellied anglerfish	Lusitanian	Uncertain	✓	Y
Lophiidae\$	<i>Lophius piscatorius</i>	Anglerfish	Lusitanian	Abundant	✓	Y
Gobiesocidae	<i>Diplecogaster bimaculata</i>	Two spotted clingfish	Lusitanian	Rare		
Gobiesocidae	<i>Lepadogaster lepadogaster</i>	Shore clingfish	Lusitanian	Status uncertain		Y
Exocoetidae	<i>Cypselurus heterurus</i>	Atlantic flying fish	Atlantic	Vagrant		
Belonidae	<i>Belone belone</i>	Garfish	Lusitanian	Rare	✓	
Scomberesocidae	<i>Scomberesox saurus</i>	Saurey pike	Atlantic	Vagrant	✓	
Atherinidae	<i>Atherina boyeri</i>	Big-scale sandsmelt	Lusitanian	Rare		
Atherinidae	<i>Atherina presbyter</i>	Sand smelt	Lusitanian	Rare	✓	

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Lampridae	<i>Lampris guttatus</i>	Opah	Atlantic	Vagrant	✓	
Trachipteridae	<i>Trachipterus arcticus</i>	Deal fish	Atlantic	Vagrant	✓	
Regalecidae	<i>Regalecus glesne</i>	Ribbon fish	Atlantic	Vagrant		
Berycidae	<i>Beryx decadactylus</i>	Beryx	Atlantic	Vagrant	✓	
Zeidae	<i>Zeus faber</i>	John dory	Lusitanian	Rare	✓	
Caproidae	<i>Capros aper</i>	Boar fish	Lusitanian	Vagrant		Y
Gasterosteidae	<i>Gasterosteus aculeatus</i>	3-spined stickleback	Boreal	Frequent	✓	
Gasterosteidae	<i>Spinachia spinachia</i>	Sea stickleback	Boreal	Rare	✓	
Centriscidae	<i>Macroramphosus scolopax</i>	Snipe-fish	Lusitanian	Vagrant	✓	
Syngnathidae\$	<i>Entelurus aequoreus</i>	Snake pipefish	Lusitanian	Rare	✓	N
Syngnathidae\$	<i>Hippocampus ramulosus</i>	Sea horse	Lusitanian	Vagrant		
Syngnathidae\$	<i>Nerophis lumbriciformis</i>	Worm pipefish	Lusitanian	Rare	✓	
Syngnathidae\$	<i>Nerophis ophidion</i>	Straight-nosed pipefish	Lusitanian	Rare	✓	
Syngnathidae\$	<i>Syngnathus acus</i>	Great pipefish	Lusitanian	Rare	✓	Y
Syngnathidae\$	<i>Syngnathus rostellatus</i>	Nilsson's pipefish	Lusitanian	Frequent	✓	N
Syngnathidae\$	<i>Syngnathus typhle</i>	Deep-snouted pipefish	Lusitanian	Rare	✓	
Sebastidae\$	<i>Helicolenus dactylopterus</i>	Blue-mouth redfish	Atlantic	Rare	✓	Y
Sebastidae\$	<i>Scorpaena scrofa</i>	Red scorpion fish	Lusitanian	Status uncertain		Y
Sebastidae\$	<i>Sebastes norvegicus</i> (=S.marinus)	Redfish	Boreal	Rare	✓	Y
Sebastidae\$	<i>Sebastes viviparus</i>	Norway haddock	Boreal	Frequent	✓	Y
Triglidae\$	<i>Aspitrigla cuculus</i>	Red gurnard	Lusitanian	Rare	✓	Y
Triglidae\$	<i>Eurigla gurnardus</i>	Grey gurnard	Lusitanian	Dominant	✓	Y
Triglidae\$	<i>Trigla lucerna</i>	Tub gurnard	Lusitanian	Moderately abundant	✓	Y
Triglidae\$	<i>Trigla lyra</i>	Piper	Lusitanian	Rare		Y
Triglidae\$	<i>Trigloporus lastoviza</i>	Streaked gurnard	Lusitanian	Rare		Y
Cottidae\$	<i>Artediiellus atlanticus</i>	Atlantic hook-ear sculpin	Boreal	Rare	✓	
Cottidae\$	<i>Icelus bicornis</i>	Two-horn sculpin	Boreal	Rare		
Cottidae\$	<i>Myoxocephalus scorpius</i>	Bullrout	Boreal	Frequent	✓	Y
Cottidae\$	<i>Micrenophrys lilljeborgii</i>	Norway bullhead	Boreal	Rare		Y
Cottidae\$	<i>Taurulus bubalis</i>	Sea scorpion	Boreal	Frequent	✓	
Cottidae	<i>Triglops murrayi</i>	Sculpin	Boreal	Rare	✓	N
Agonidae	<i>Agonus cataphractus</i>	Pogge (armed bullhead)	Boreal	Rare	✓	Y
Cyclopteridae	<i>Cyclopterus lumpus</i>	Lumpsucker	Boreal	Rare	✓	Y
Liparidae	<i>Liparis liparis</i>	Sea snail	Boreal	Rare	✓	Y
Liparidae	<i>Liparis montagui</i>	Montague's seasnail	Boreal	Rare	✓	N

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Percichthyidae	<i>Dicentrarchus labrax</i>	European seabass	Lusitanian	Rare	✓	
Percichthyidae	<i>Polyprion americanus</i>	Wreck-fish	Lusitanian	Vagrant		
Serranidae	<i>Serranus cabrilla</i>	Comber	Lusitanian	Vagrant		
Carangidae	<i>Naucrates ductor</i>	Pilot fish	Atlantic	Vagrant		
Carangidae	<i>Trachinotus ovatus</i>	Derbio	Lusitanian	Vagrant	✓	
Carangidae	<i>Trachurus trachurus</i>	Horse-mackerel	Lusitanian	Dominant	✓	
Coryphaenidae	<i>Coryphaena equiselis</i>	Pompano dolphinfish	Circumglobal	Vagrant		
Bramidae	<i>Brama brama</i>	Rays bream	Atlantic	Moderately abundant	✓	
Bramidae	<i>Pterycombus brama</i>	Silver pomfret	Atlantic	Vagrant		
Bramidae	<i>Taractes asper</i>	Rough pomfret	Atlantic	Vagrant		
Bramidae	<i>Taractichthys longipinnis</i>	Long-finned bream	Atlantic	Vagrant		
Sparidae	<i>Boops boops</i>	Bogue	Lusitanian	Vagrant	✓	
Sparidae	<i>Dentex dentex</i>	Dentex	Lusitanian	Vagrant		
Sparidae	<i>Dentex maroccanus</i>	Morocco dentex	Lusitanian	Vagrant		
Sparidae	<i>Pagellus acarne</i>	Auxillary seabream	Lusitanian	Vagrant	✓	
Sparidae	<i>Pagellus bogaraveo</i>	Red seabream	Lusitanian	Rare	✓	
Sparidae	<i>Pagellus erythrinus</i>	Pandora	Lusitanian	Vagrant	✓	Y
Sparidae	<i>Sarpa salpa</i>	Saupe	Lusitanian	Vagrant		
Sparidae	<i>Sparus aurata</i>	Gilt-head seabream	Lusitanian	Vagrant	✓	
Sparidae	<i>Spondylisoma cantharus</i>	Black seabream	Lusitanian	Rare	✓	Y
Sciaenidae	<i>Argyrosomus regius</i>	Meagre	Lusitanian	Vagrant	✓	
Mullidae	<i>Mullus barbatus</i>	Red mullet	Lusitanian	Vagrant	✓	N
Mullidae	<i>Mullus surmuletus</i>	Striped red mullet	Lusitanian	Rare	✓	Y
Cepolidae	<i>Cepola rubescens</i>	Red bandfish	Lusitanian	Vagrant	✓	Y
Mugilidae\$	<i>Chelon labrosus</i>	Thick lipped mullet	Lusitanian	Frequent	✓	N
Mugilidae\$	<i>Liza aurata</i>	Golden mullet	Lusitanian	Rare	✓	
Mugilidae\$	<i>Liza ramado</i>	Thin lipped mullet	Lusitanian	Vagrant	✓	
Mugilidae\$	<i>Mugil cephalus</i>	Flathead mullet		Status uncertain		Y
Labridae\$	<i>Acantholabrus palloni</i>	Scale-rayed wrasse	Lusitanian	Rare		
Labridae\$	<i>Centrolabrus exoletus</i>	Small-mouthed wrasse	Lusitanian	Rare	✓	
Labridae\$	<i>Coris julis</i>	Rainbow wrasse	Lusitanian		✓	
Labridae\$	<i>Symphodus bailloni</i>	Baillon's wrasse	Lusitanian			Y
Labridae\$	<i>Symphodus melops</i>	Corkwing	Lusitanian	Rare	✓	
Labridae\$	<i>Ctenolabrus rupestris</i>	Goldsinny	Lusitanian	Rare	✓	Y
Labridae\$	<i>Labrus bergylta</i>	Ballan wrasse	Lusitanian	Rare	✓	N

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Labridae\$	<i>Labrus mixtus</i>	Cuckoo wrasse	Lusitanian	Rare	✓	Y
Zoarcidae\$	<i>Lycenchelys sarsii</i>	Sar's eelpout	Boreal	Rare		N
Zoarcidae\$	<i>Lycodes esmarkii</i>	Esmarl's eelpout	Boreal	Rare		N
Zoarcidae\$	<i>Lycodes vahlii</i>	Vahl's eelpout	Boreal	Rare	✓	Y
Zoarcidae\$	<i>Zoarces viviparus</i>	Eelpout	Boreal	Frequent	✓	Y
Stichaeidae\$	<i>Chirolophis ascanii</i>	Yarrell's blenny	Boreal	Rare	✓	Y
Stichaeidae\$	<i>Leptoclinus maculatus</i>	Spotted snake blenny	Boreal	Rare	✓	N
Stichaeidae\$	<i>Lumpenus lampretaeformis</i>	Snake blenny	Boreal	Rare	✓	Y
Pholidae	<i>Pholis gunnellus</i>	Butter fish	Boreal	Rare	✓	Y
Anarhichadidae	<i>Anarhichas denticulatus</i>	Jelly cat	Boreal	Vagrant		
Anarhichadidae	<i>Anarhichas lupus</i>	Wolf fish	Boreal	Moderately abundant	✓	Y
Anarhichadidae	<i>Anarhichas minor</i>	Spotted catfish	Boreal	Rare		N
Trachinidae\$	<i>Echiichthys vipera</i>	Lesser weever fish	Lusitanian	Moderately abundant	✓	Y
Trachinidae\$	<i>Trachinus draco</i>	Greater weever fish	Lusitanian	Rare	✓	Y
Blenniidae\$	<i>Lipophrys pholis</i>	Shanny	Lusitanian	Vagrant	✓	
Blenniidae\$	<i>Parablennius gattorugine</i>	Tompot blenny	Lusitanian	Vagrant		
Ammodytidae	<i>Ammodytes marinus</i>	Sandeel	Boreal	Dominant	✓	
Ammodytidae	<i>Ammodytes tobianus</i>	Sandeel	Boreal	Frequent	✓	
Ammodytidae	<i>Gymnammodytes semisquamatus</i>	Smooth sandeel	Lusitanian	Moderately abundant	✓	
Ammodytidae	<i>Hyperoplus immaculatus</i>	Immaculate sandeel	Lusitanian	Frequent		
Ammodytidae	<i>Hyperoplus lanceolatus</i>	Great sandeel	Boreal	Abundant	✓	
Callionymidae\$	<i>Callionymus lyra</i>	Common dragonet	Lusitanian	Moderately abundant	✓	Y
Callionymidae\$	<i>Callionymus maculatus</i>	Spotted dragonet	Lusitanian	Rare	✓	Y
Callionymidae\$	<i>Callionymus reticulatus</i>	Reticulate dragonet	Lusitanian	Rare	✓	N
Gobiidae\$	<i>Aphia minuta</i>	Transparent goby	Lusitanian	Frequent	✓	
Gobiidae\$	<i>Buenia jeffreysii</i>	Jeffrey's goby	Boreal	Rare	✓	
Gobiidae\$	<i>Crystalllogobius linearis</i>	Crystal goby	Lusitanian	Frequent	✓	N
Gobiidae\$	<i>Gobius couchi</i>	Couch's goby		Status uncertain		Y
Gobiidae\$	<i>Gobius niger</i>	Black goby	Lusitanian	Frequent	✓	N
Gobiidae\$	<i>Gobiusculus flavescens</i>	Two-spot goby	Lusitanian	Rare	✓	
Gobiidae\$	<i>Lebetus guilleti</i>	Guillet's goby	Boreal	Rare		
Gobiidae\$	<i>Lebetus scorpioides</i>	Diminutive goby	Boreal	Frequent		
Gobiidae\$	<i>Lesueurigobius friesii</i>	Fries's goby	Boreal	Frequent	✓	Y
Gobiidae\$	<i>Pomatoschistus lozanoi</i>	Lozano's goby		?	✓	
Gobiidae\$	<i>Pomatoschistus microps</i>	Common goby	Lusitanian	Frequent	✓	N

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Gobiidae\$	<i>Pomatoschistus minutus</i>	Sand goby	Lusitanian	Frequent	✓	N
Gobiidae\$	<i>Pomatoschistus norvegicus</i>	Norwegian goby	Lusitanian	Rare		
Gobiidae\$	<i>Pomatoschistus pictus</i>	Painted goby	Lusitanian	Rare	✓	
Gobiidae\$	<i>Thorogobius ephippiatus</i>	Leopard-spotted goby	Lusitanian	Rare	✓	
Trichiuridae	<i>Aphanopus carbo</i>	Black scabbardfish	Atlantic	Vagrant		
Trichiuridae	<i>Lepidopus caudatus</i>	Scabbard fish	Atlantic	Vagrant		
Scombridae	<i>Auxis rochei</i>	Frigate mackerel	Atlantic	Vagrant		
Scombridae	<i>Euthynnus alletteratus</i>	Little tunny	Atlantic	Vagrant	✓	
Scombridae	<i>Katsuwonus pelamis</i>	Skipjack tuna	Atlantic	Vagrant	✓	
Scombridae	<i>Orcynopsis unicolor</i>	Plain bonito	Atlantic	Vagrant	✓	
Scombridae	<i>Sarda sarda</i>	Bonito	Atlantic	Vagrant	✓	
Scombridae	<i>Scomber scombrus</i>	Atlantic mackerel	Atlantic	Dominant	✓	
Scombridae	<i>Thunnus alalunga</i>	Albacore	Atlantic	Vagrant		
Scombridae	<i>Thunnus thynnus</i>	Blue-fin tunny	Atlantic	Rare	✓	
Xiphiidae	<i>Xiphias gladius</i>	Swordfish	Atlantic	Vagrant	✓	
Luvaridae	<i>Luvarus imperialis</i>	Luvar	Atlantic	Vagrant		
Centrolophidae	<i>Centrolophus niger</i>	Blackfish	Atlantic	Vagrant		
Centrolophidae	<i>Schedophilus medusophagus</i>	Cornish blackfish	Atlantic	Vagrant		
Scophthalmidae	<i>Lepidorhombus boscii</i>	Fourspotted megrim	Lusitanian	Rare		Y
Scophthalmidae	<i>Lepidorhombus whiffiagonis</i>	Megrim	Lusitanian	Abundant	✓	Y
Scophthalmidae\$	<i>Phrynorhombus norvegicus</i>	Norwegian topknot	Boreal	Rare	✓	Y
Scophthalmidae	<i>Psetta maxima</i>	Turbot	Lusitanian	Moderately abundant	✓	Y
Scophthalmidae	<i>Scophthalmus rhombus</i>	Brill	Lusitanian	Frequent	✓	Y
Scophthalmidae\$	<i>Zeugopterus punctatus</i>	Topknot	Lusitanian	Rare	✓	Y
Bothidae	<i>Arnoglossus imperialis</i>	Imperial scald fish	Lusitanian	Status uncertain		N
Bothidae	<i>Arnoglossus laterna</i>	Scald fish	Lusitanian	Rare	✓	N
Pleuronectidae	<i>Glyptocephalus cynoglossus</i>	Witch	Boreal	Moderately abundant	✓	Y
Pleuronectidae	<i>Hippoglossoides platessoides</i>	Long-rough dab	Boreal	Dominant	✓	Y
Pleuronectidae	<i>Hippoglossus hippoglossus</i>	Halibut	Boreal	Frequent	✓	Y
Pleuronectidae	<i>Limanda limanda</i>	Dab	Boreal	Dominant	✓	Y
Pleuronectidae	<i>Microstomus kitt</i>	Lemon sole	Boreal	Abundant	✓	Y
Pleuronectidae	<i>Platichthys flesus</i>	Flounder	Lusitanian	Frequent	✓	Y
Pleuronectidae	<i>Pleuronectes platessa</i>	European plaice	Lusitanian	Dominant	✓	Y
Pleuronectidae	<i>Reinhardtius hippoglossoides</i>	Greenland halibut	Boreal	Rare		
Soleidae\$	<i>Buglossidium luteum</i>	Solenette	Lusitanian	Rare	✓	Y

Family	Species	Common name	Biogeography	Occurrence*	FishBase	Length-weight
Soleidae\$	<i>Microchirus variegatus</i>	Thickback sole	Lusitanian	Rare	✓	Y
Soleidae\$	<i>Solea lascaris</i>	Sand sole	Lusitanian	Vagrant		Y
Soleidae\$	<i>Solea solea</i>	Sole	Lusitanian	Abundant	✓	Y
Balistidae	<i>Balistes capriscus</i>	Grey trigger fish	Lusitanian	Vagrant		Y
Tetraodontidae	<i>Lagocephalus lagocephalus</i>	Puffer fish	Atlantic	Vagrant		
Molidae	<i>Mola mola</i>	Sunfish	Atlantic	Vagrant	✓	
Molidae	<i>Ranzania laevis</i>	Truncated sunfish	Atlantic	Vagrant		

Table 7.2.2. Checklist of fish species that occur in the estuaries feeding into the North Sea (NB: Diadromous and estuarine species that are frequently recorded in fully marine environments are listed in the previous Table 7.2.1)

Family	Scientific name	Common name
Cyprinidae	<i>Abramis brama</i>	Carp bream
Cyprinidae	<i>Aspius aspius</i>	Asp
Cyprinidae	<i>Barbus barbus</i>	Barbel
Cyprinidae	<i>Carassius carassius</i>	Crucian carp
Cyprinidae	<i>Gobio gobio</i>	Gudgeon
Cyprinidae	<i>Leuciscus cephalus</i>	European chub
Cyprinidae	<i>Leuciscus idus</i>	Ide
Cyprinidae	<i>Rutilus rutilus</i>	Roach
Cyprinidae	<i>Scardinius erythrophthalmus</i>	Rudd
Cyprinidae	<i>Vimba vimba</i>	Baltic vimba
Cobitidae	<i>Cobitis taenia</i>	Spined loach
Esocidae	<i>Esox lucius</i>	Pike
Lotidae	<i>Lota lota</i>	Burbot
Gasterosteidae	<i>Pungitius pungitius</i>	Ninespine stickleback
Cottidae	<i>Cottus gobio</i>	Bullhead
Percidae	<i>Gymnocephalus cernuus</i>	Ruffe
Percidae	<i>Perca fluviatilis</i>	European perch
Percidae	<i>Sander lucioperca</i>	Zander

Table 7.3.2.1. The difference in area between ICES area IV and the IBTS area and the raising factor used to make them comparable.

Area	ICES area Km ²	IBTS area Km ²	Raising Factor
IVa1	133,049	100,900	1.3174468
IVa2	131,294	85,069	1.543383
IVb1	125,519	121,142	1.0361325
IVb2	151,764	143,427	1.0581291
IVc	66,572	46,655	1.4269104

Table 7.3.3.1. Numbers of all demersal fish and numbers of the nine selected species (and percentage contribution) sampled each year in the IBTS in each year.

Year	All demersal fish	Nine species	Percentage contribution
1998	15,376,053	14,952,958	97.2
1999	47,984,891	47,094,074	98.1
2000	28,254,466	27,484,404	97.3
2001	18,062,565	17,519,262	97.0
2002	16,208,788	14,747,917	91.0
2003	12,140,167	11,234,959	92.5

Table 7.3.3.2. Raised area weighted catch (tonnes) for each of the nine demersal fish species in each of the 5 areas 1998 – 2003, HAD (haddock), WHI (whiting), COD (Cod), NPO (Norway pout), GGU (grey gurnard), PLA (plaice), LSO (lemon sole), CDA (common dab), LRD (long rough dab), and the raising factor (RF) used.

SP	AREA	IBTS survey						RF	Raised North Sea					
		1998	1999	2000	2001	2002	2003		1998	1999	2000	2001	2002	2003
HAD	IVa1	107,498	171,662	279,287	219,713	202,633	215,541	1.3174	141,623	226,155	367,946	289,460	266,958	283,964
	IVa2	48,942	181,174	150,882	177,537	172,204	77,156	1.5434	75,536	279,621	232,868	274,008	265,777	119,081
	IVb1	128,128	129,227	344,137	274,368	231,290	145,562	1.0361	132,758	133,896	356,571	284,282	239,647	150,822
	IVb2	18,659	19,270	101,722	26,749	20,813	77,656	1.0581	19,744	20,390	107,635	28,304	22,023	82,170
	IVc	17	18	277	1,098	0	0	1.4269	24	26	395	1,566	0	0
	TOTAL	303,245	501,351	876,304	699,465	626,940	515,915		369,685	660,089	1,065,416	877,620	794,405	636,037
WHI	IVa1	78,500	160,544	120,935	70,190	86,931	80,988	1.3174	103,420	211,508	159,325	92,471	114,528	106,697
	IVa2	13,190	30,711	23,688	27,469	20,632	16,892	1.5434	20,358	47,399	36,560	42,396	31,843	26,071
	IVb1	146,378	218,397	292,977	236,953	231,912	195,413	1.0361	151,667	226,288	303,563	245,514	240,292	202,474
	IVb2	52,874	168,185	137,724	221,954	106,985	75,376	1.0581	55,947	177,961	145,730	234,856	113,203	79,757
	IVc	38,872	18,177	61,795	91,203	19,647	8,754	1.4269	55,468	25,937	88,176	130,138	28,035	12,491
	TOTAL	329,815	596,014	637,118	647,769	466,107	377,423		386,860	689,094	733,353	745,376	527,900	427,491
COD	IVa1	8,596	4,335	4,549	3,934	6,111	4,726	1.3174	11,324	5,711	5,993	5,183	8,050	6,226
	IVa2	16,114	12,520	11,701	15,041	8,109	6,450	1.5434	24,871	19,323	18,058	23,214	12,515	9,955
	IVb1	14,446	6,052	5,046	2,544	3,698	4,753	1.0361	14,968	6,271	5,228	2,635	3,832	4,925
	IVb2	20,908	11,133	7,287	24,165	7,240	1,391	1.0581	22,123	11,780	7,710	25,570	7,661	1,472
	IVc	6,537	3,073	3,973	4,645	9,374	427	1.4269	9,328	4,385	5,669	6,628	13,375	610
	TOTAL	66,601	37,113	32,556	50,329	34,531	17,747		82,615	47,470	42,660	63,230	45,433	23,187
NPO	IVa1	60,956	75,336	207,808	79,433	70,834	46,258	1.3174	80,307	99,251	273,776	104,648	93,320	60,943
	IVa2	36,603	94,246	101,272	57,793	20,951	16,382	1.5434	56,492	145,458	156,302	89,197	32,335	25,283
	IVb1	1,231	20,326	97,266	16,129	4,301	2,076	1.0361	1,275	21,060	100,780	16,711	4,456	2,151
	IVb2	4,096	674	1,323	1,735	1	81	1.0581	4,334	714	1,400	1,835	1	85
	IVc	0	0	0	35	0	0	1.4269	0	0	0	50	0	0
	TOTAL	102,886	190,583	407,669	155,124	96,086	64,796		142,408	266,483	532,258	212,443	130,112	88,462
GGU	IVa1	2,959	3,875	2,357	2,440	2,685	4,002	1.3174	3,898	5,105	3,105	3,215	3,537	5,273
	IVa2	972	2,295	2,240	4,057	3,531	4,136	1.5434	1,501	3,542	3,457	6,261	5,450	6,383
	IVb1	30,259	35,787	29,518	19,896	32,111	28,503	1.0361	31,352	37,080	30,585	20,615	33,272	29,533
	IVb2	48,465	52,836	39,765	56,254	33,727	36,024	1.0581	51,283	55,907	42,077	59,524	35,687	38,118
	IVc	1,402	1,041	459	2,397	496	23	1.4269	2,001	1,485	655	3,420	708	33
	TOTAL	84,057	95,834	74,339	85,043	72,551	72,688		90,034	103,120	79,878	93,034	78,654	79,339
PLA	IVa1	662	1,437	829	2,077	1,558	1,841	1.3174	872	1,893	1,092	2,736	2,053	2,425
	IVa2	115	299	293	493	543	520	1.5434	177	462	453	760	838	802
	IVb1	2,709	2,170	3,335	3,717	5,585	4,904	1.0361	2,807	2,248	3,455	3,851	5,786	5,082
	IVb2	17,997	20,306	8,412	11,467	10,872	14,400	1.0581	19,043	21,487	8,901	12,134	11,504	15,237
	IVc	939	3,140	1,050	2,257	2,099	1,160	1.4269	1,341	4,481	1,498	3,220	2,996	1,655
	TOTAL	22,422	27,353	13,919	20,010	20,658	22,824		24,240	30,571	15,399	22,702	23,178	25,201
LSO	IVa1	7,017	7,145	10,527	10,032	6,495	8,610	1.3174	9,245	9,413	13,868	13,217	8,557	11,344
	IVa2	1,092	1,817	1,991	1,915	2,083	1,713	1.5434	1,686	2,805	3,073	2,955	3,215	2,644
	IVb1	7,426	8,226	7,099	5,095	7,329	7,496	1.0361	7,695	8,523	7,355	5,279	7,594	7,766
	IVb2	3,800	2,132	1,850	1,374	1,925	2,539	1.0581	4,021	2,256	1,957	1,454	2,037	2,686
	IVc	230	116	55	195	155	0	1.4269	328	166	79	279	221	0
	TOTAL	19,566	19,437	21,522	18,611	17,988	20,358		22,974	23,163	26,333	23,183	21,624	24,441
CDA	IVa1	3,583	7,369	5,509	4,718	4,896	2,494	1.3174	4,720	9,709	7,257	6,215	6,450	3,286
	IVa2	2,663	8,034	4,247	6,922	5,048	5,181	1.5434	4,110	12,400	6,554	10,684	7,792	7,996
	IVb1	30,108	44,235	32,649	43,828	67,708	40,642	1.0361	31,196	45,833	33,829	45,412	70,155	42,110
	IVb2	98,250	82,311	72,240	118,451	143,598	153,965	1.0581	103,961	87,096	76,440	125,337	151,945	162,915
	IVc	7,930	7,260	4,484	35,032	10,249	5,119	1.4269	11,315	10,359	6,398	49,988	14,625	7,305
	TOTAL	142,534	149,209	119,129	208,952	231,500	207,401		155,303	165,396	130,478	237,636	250,966	223,611
LRD	IVa1	1,505	3,950	3,017	2,073	2,169	3,525	1.3174	1,983	5,204	3,974	2,731	2,857	4,644
	IVa2	2,789	5,133	4,412	4,430	2,495	3,684	1.5434	4,304	7,922	6,809	6,838	3,851	5,686
	IVb1	4,344	9,513	8,697	4,499	10,190	7,127	1.0361	4,500	9,857	9,011	4,661	10,558	7,384
	IVb2	13,248	6,811	5,330	9,332	6,893	7,816	1.0581	14,018	7,206	5,640	9,874	7,294	8,271
	IVc	14	31	0	10	0	0	1.4269	20	44	0	14	0	0
	TOTAL	21,899	25,437	21,455	20,344	21,747	22,152		24,826	30,233	25,434	24,118	24,560	25,985

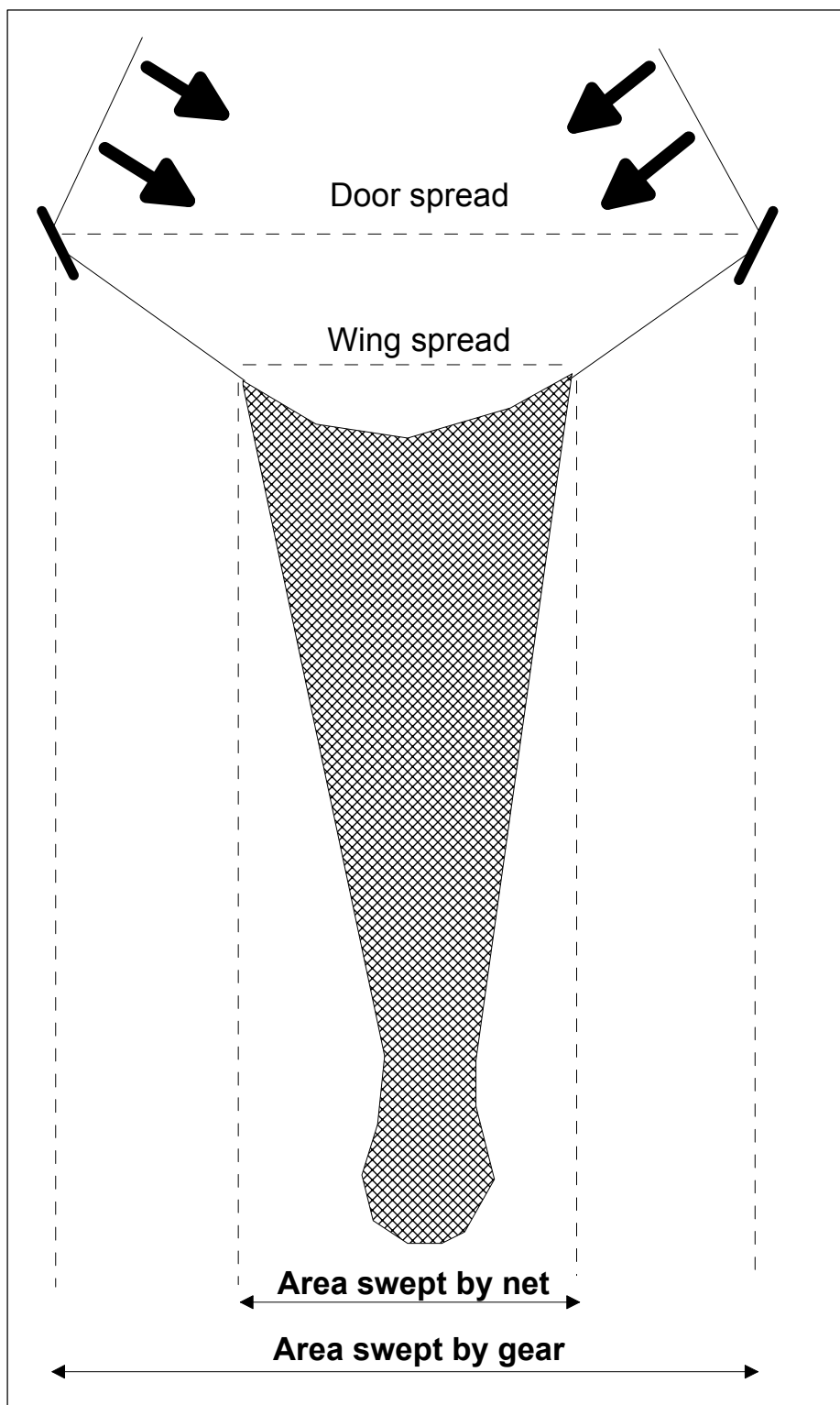


Figure 7.3.2.1. Schematic drawing of a fishing net illustrating the terms “wing-spread” and “door-spread”. Arrows indicate the possible herding effect of the otterboards and sweeps.

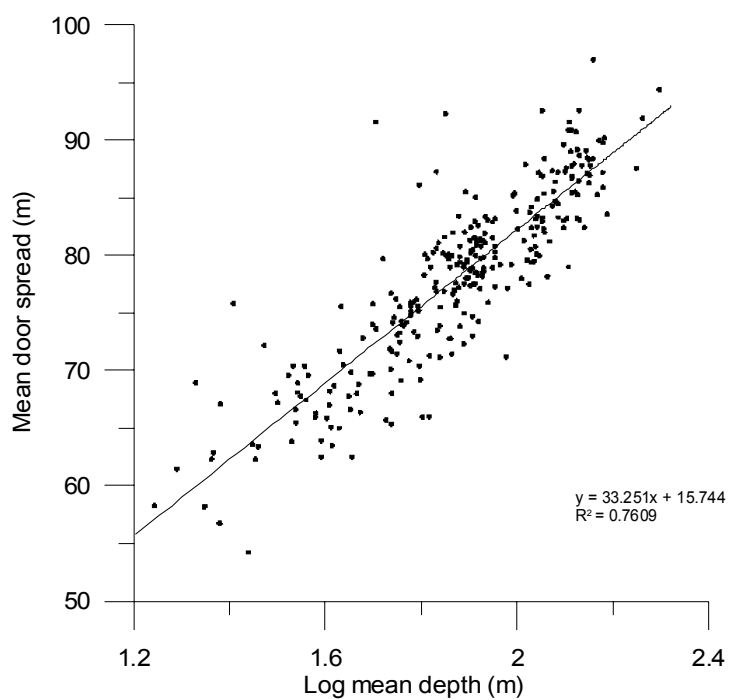
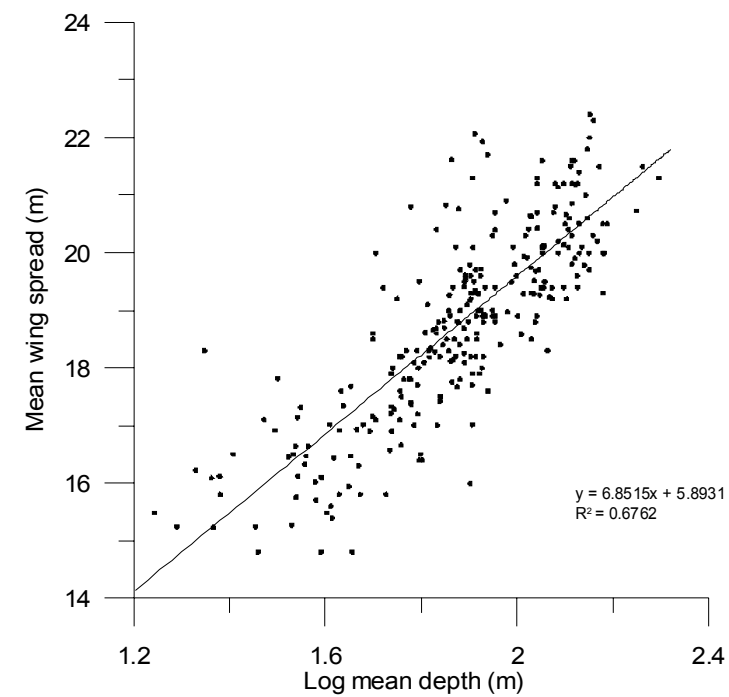


Figure 7.3.2.2. Relationship between mean wing and door spread and log depth using SCANMAR© data collected on the Scottish 3rd Quarter IBTS.

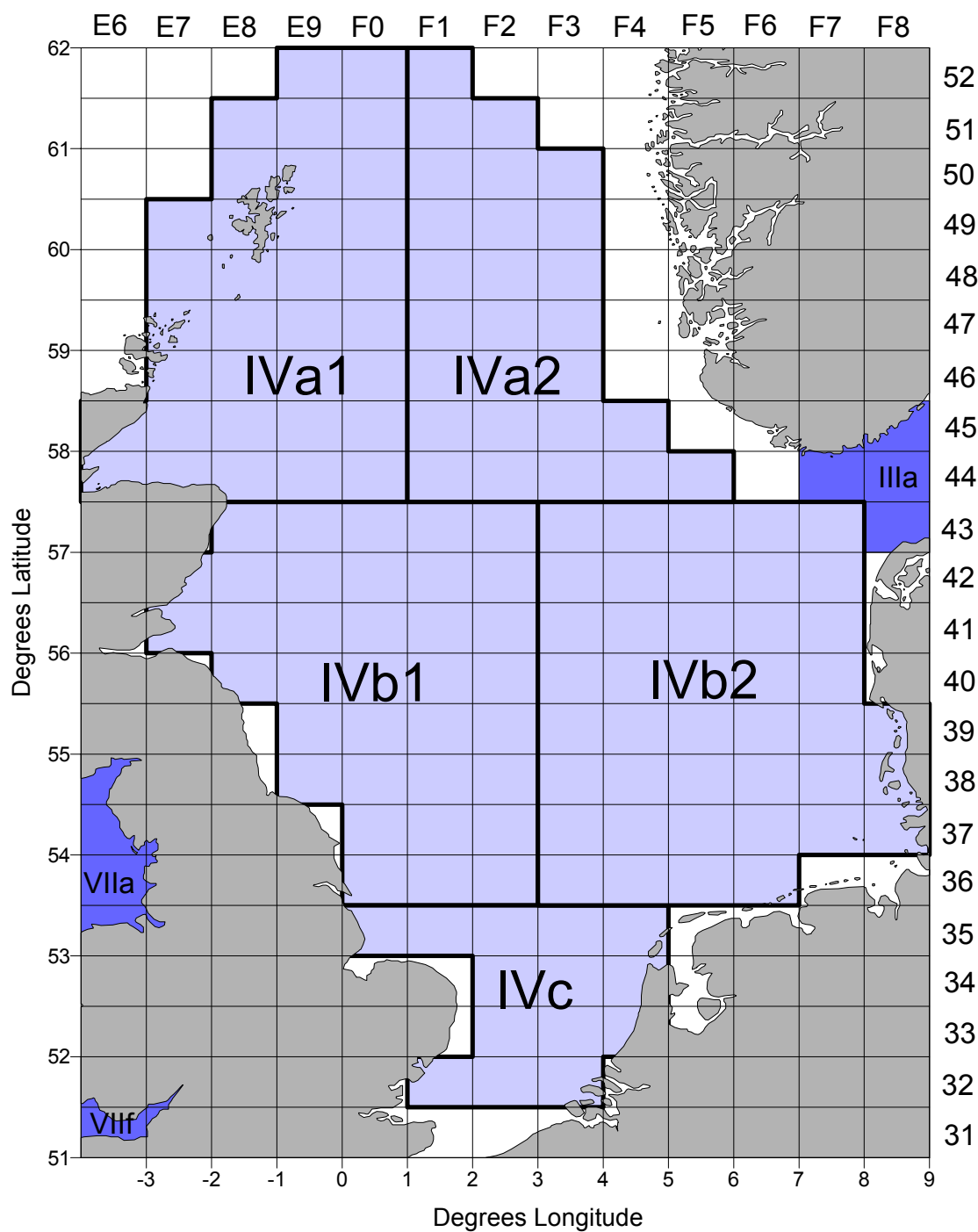


Figure 7.3.2.3. Areas shaded light blue are parts of ICES area IV which are included in the IBTS study area. White areas are part of ICES area IV which is not included in the IBTS study areas. The IBTS area has been divided into five parts, area IVa1, IVa2, IVb1 and IVc.

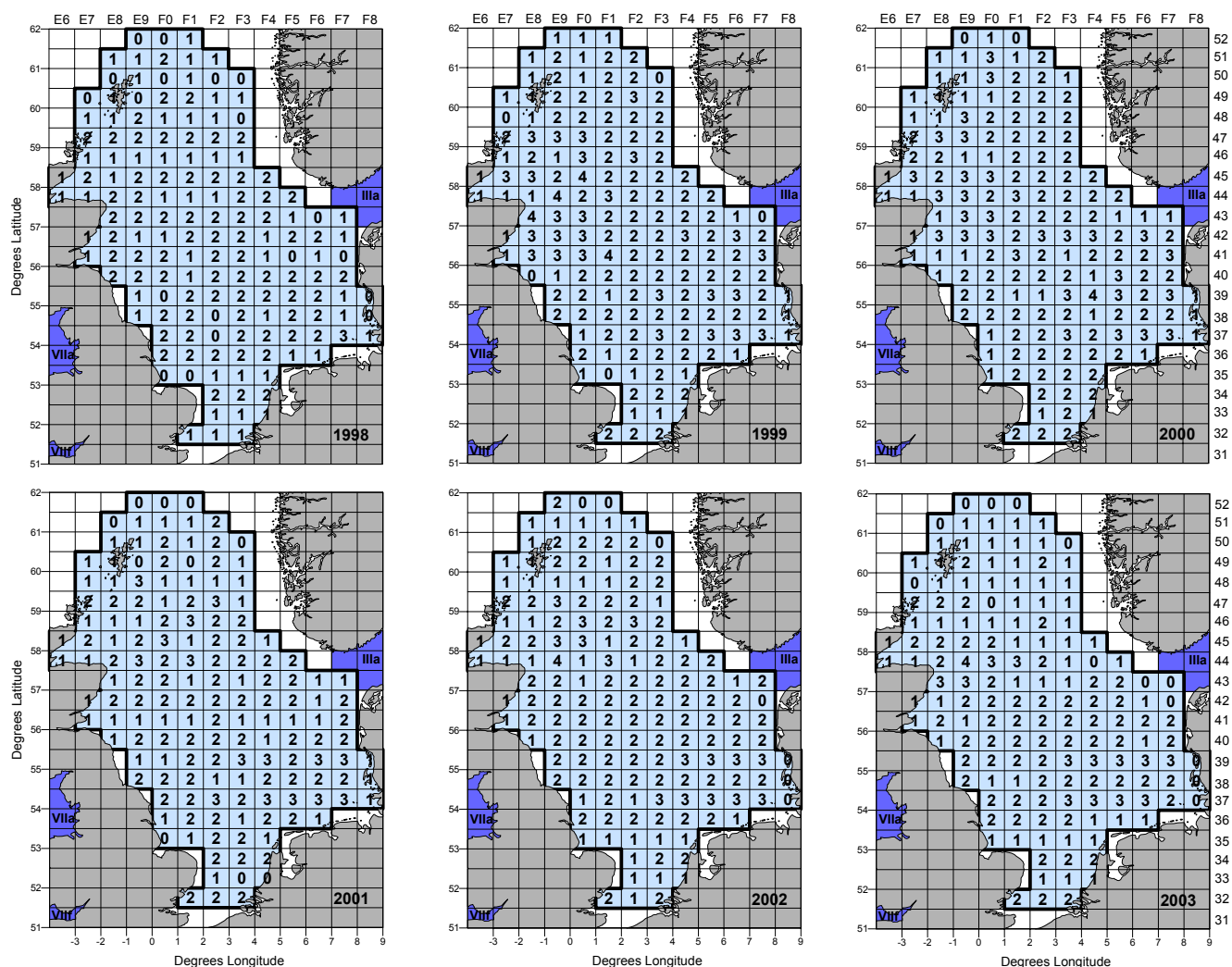


Figure 7.3.2.4. Number of hauls in each statistical rectangle, 1998 – 2003.

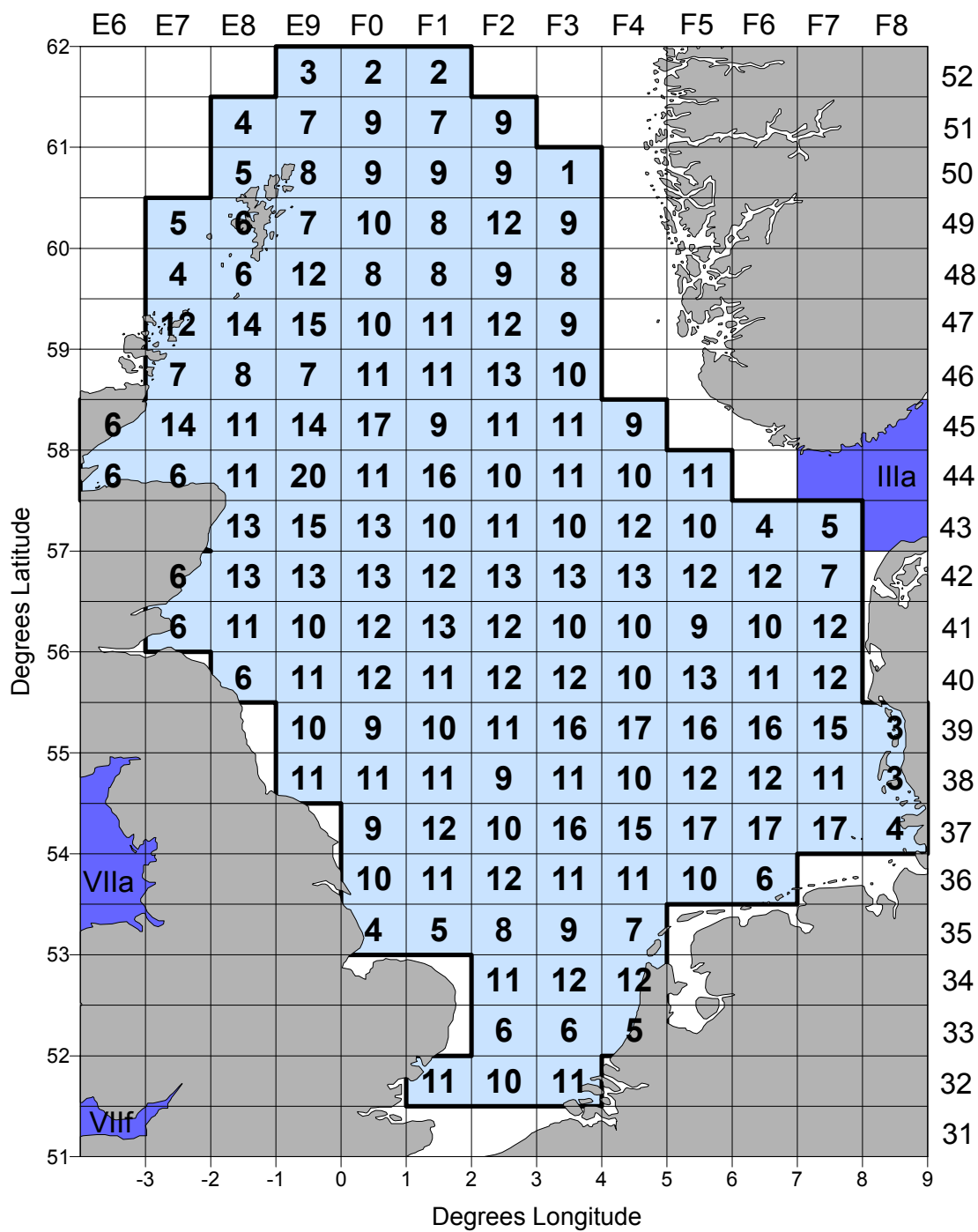


Figure 7.3.2.5. Total number of times each statistical rectangle was fished over the six year period 1998 – 2003.

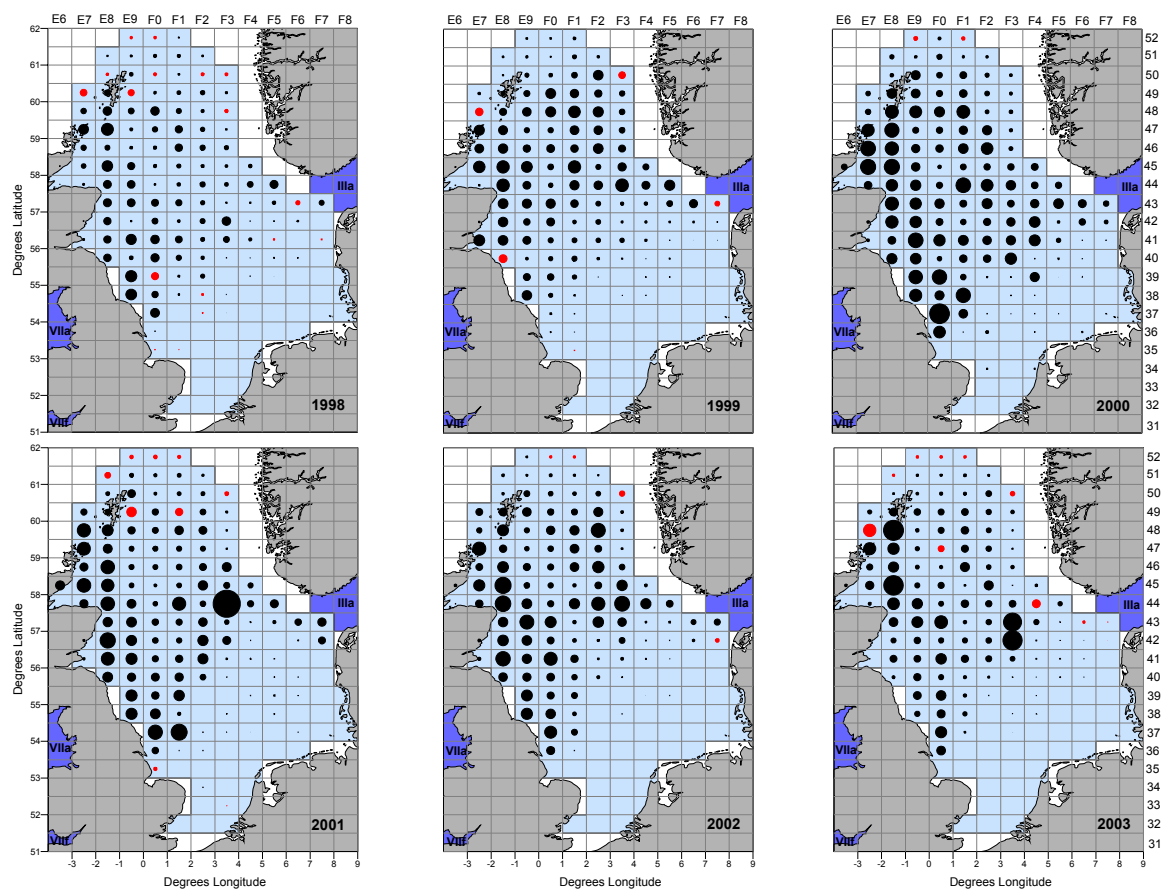


Figure 7.3.3.1. The distribution and estimated area-weighted catch of haddock in each statistical rectangle, 1998–2003 (Red symbols indicate interpolated data). Symbol size varies from 0 – 76,000 tonnes and is the same for all years.

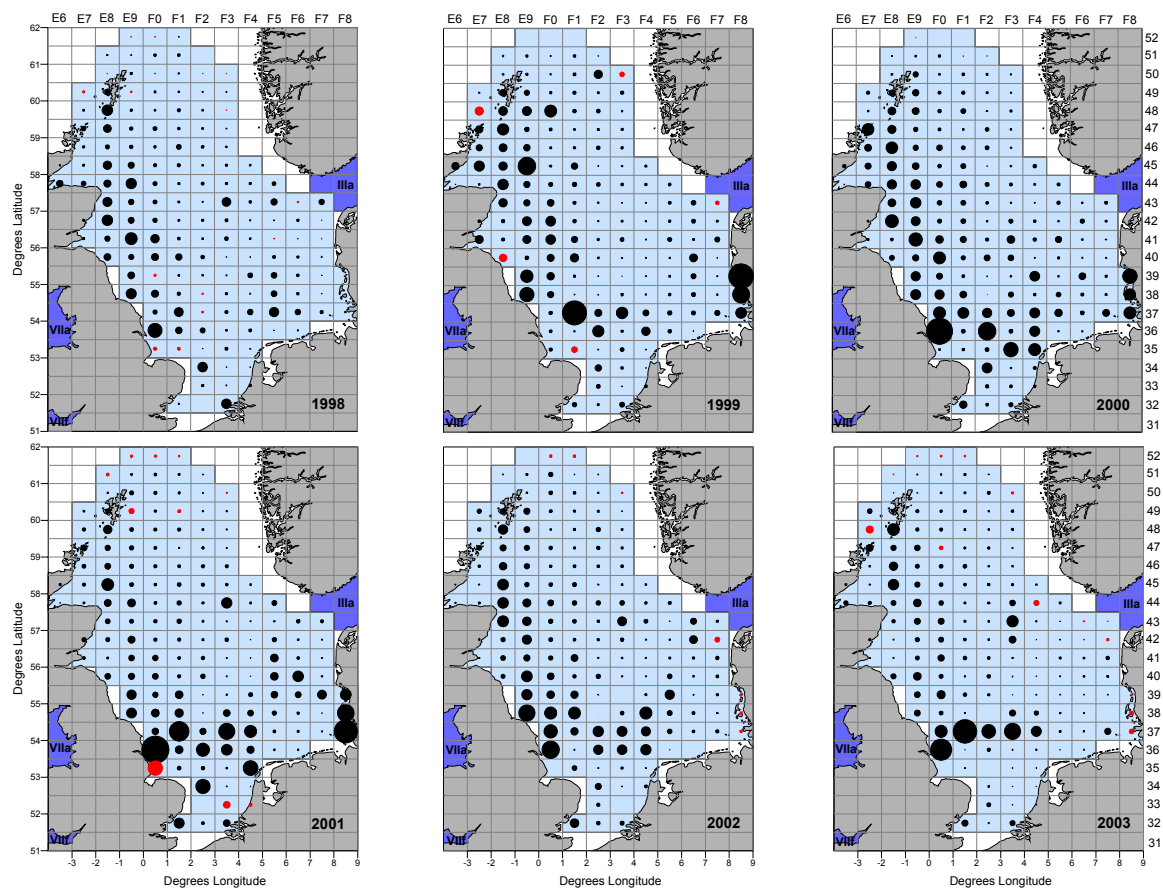


Figure 7.3.3.2. The distribution and estimated area-weighted catch of whiting in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 79,000 tonnes and is the same for all years.

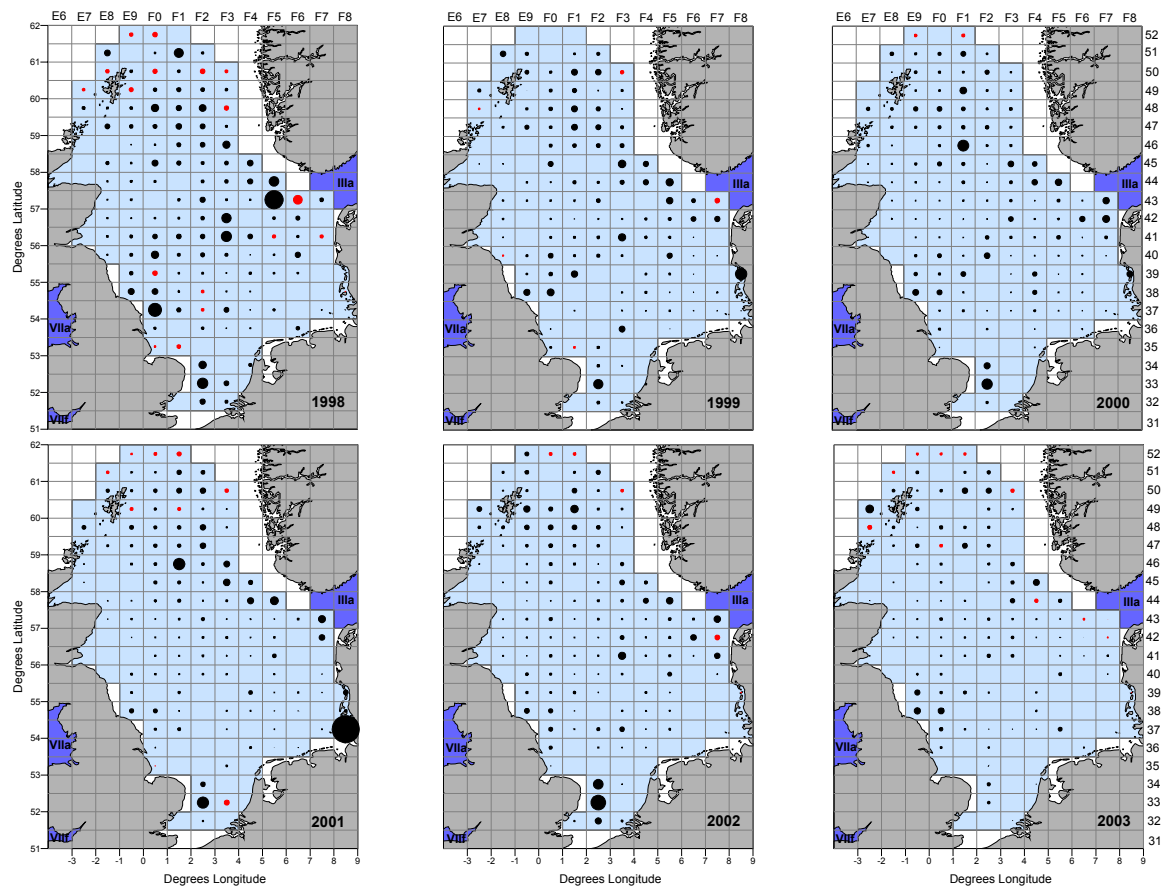


Figure 7.3.3.3. The distribution and estimated area-weighted catch of cod in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 20,000 tonnes and is the same for all years.

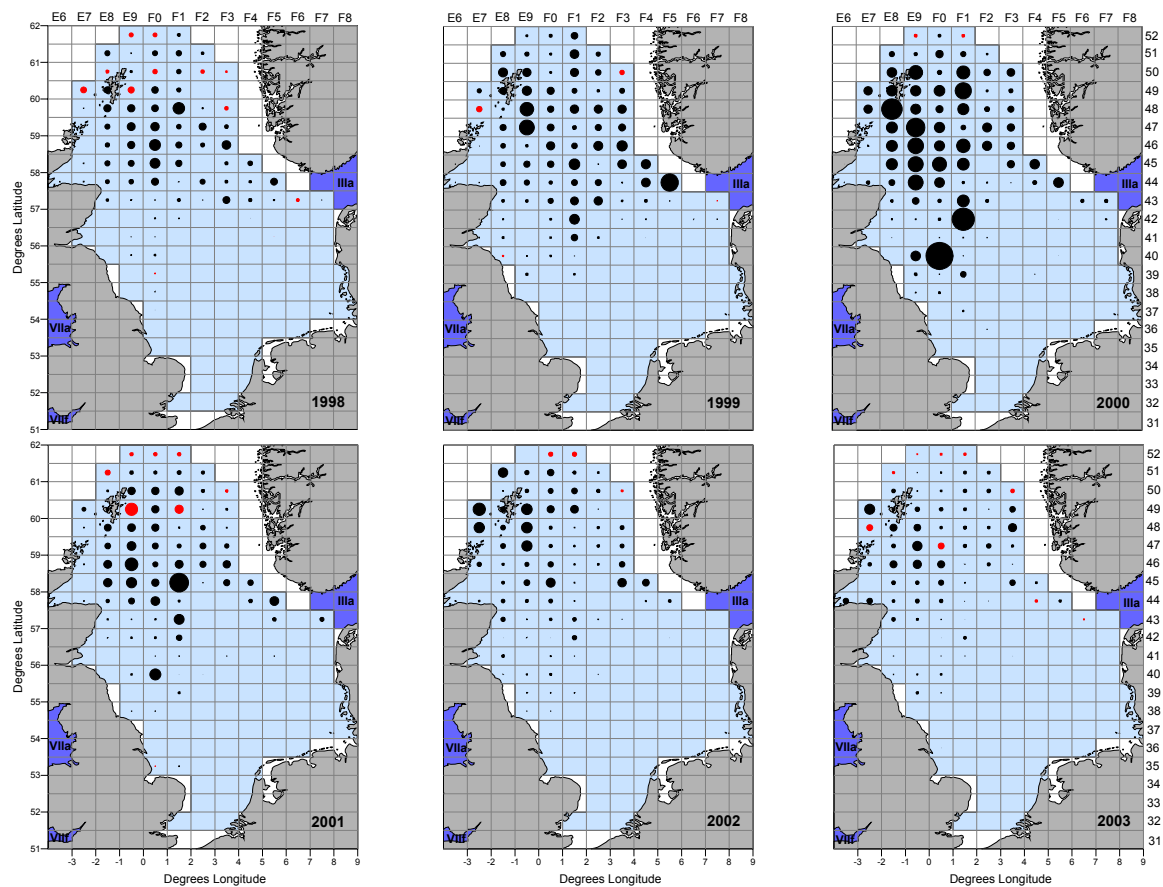


Figure 7.3.3.4. The distribution and estimated area-weighted catch of Norway pout in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 45,000 tonnes and is the same for all years.

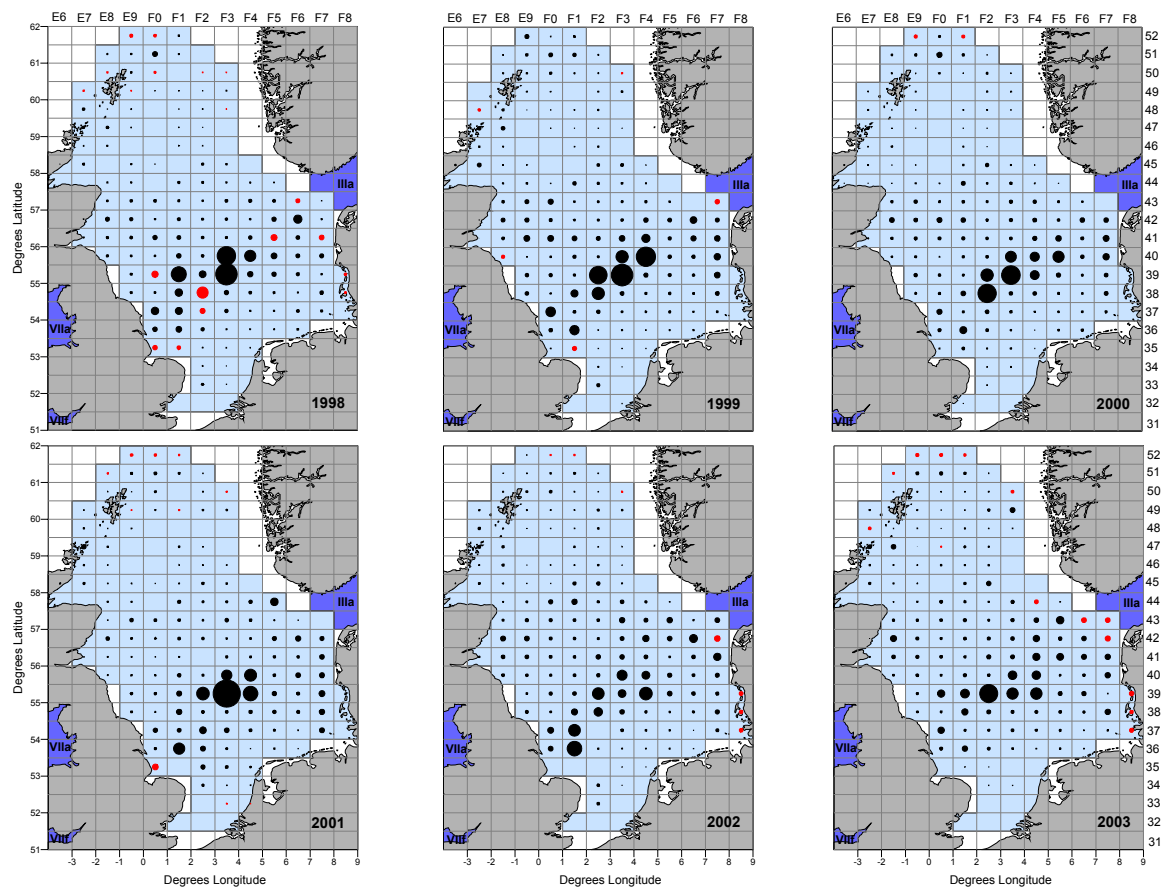


Figure 7.3.3.5. The distribution and estimated area-weighted catch of grey gurnard in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 27,000 tonnes and is the same for all years.

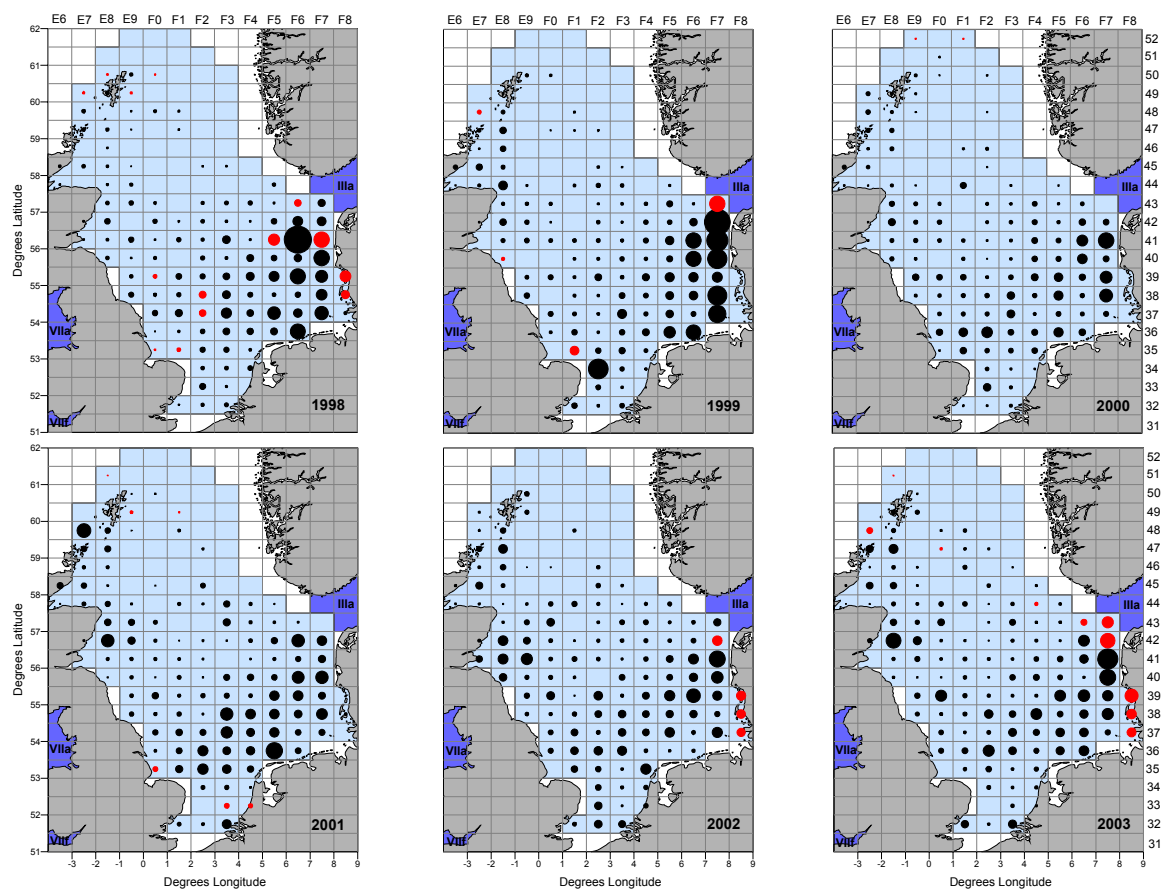


Figure 7.3.3.6. The distribution and estimated area-weighted catch of plaice in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 3,500 tonnes and is the same for all years.

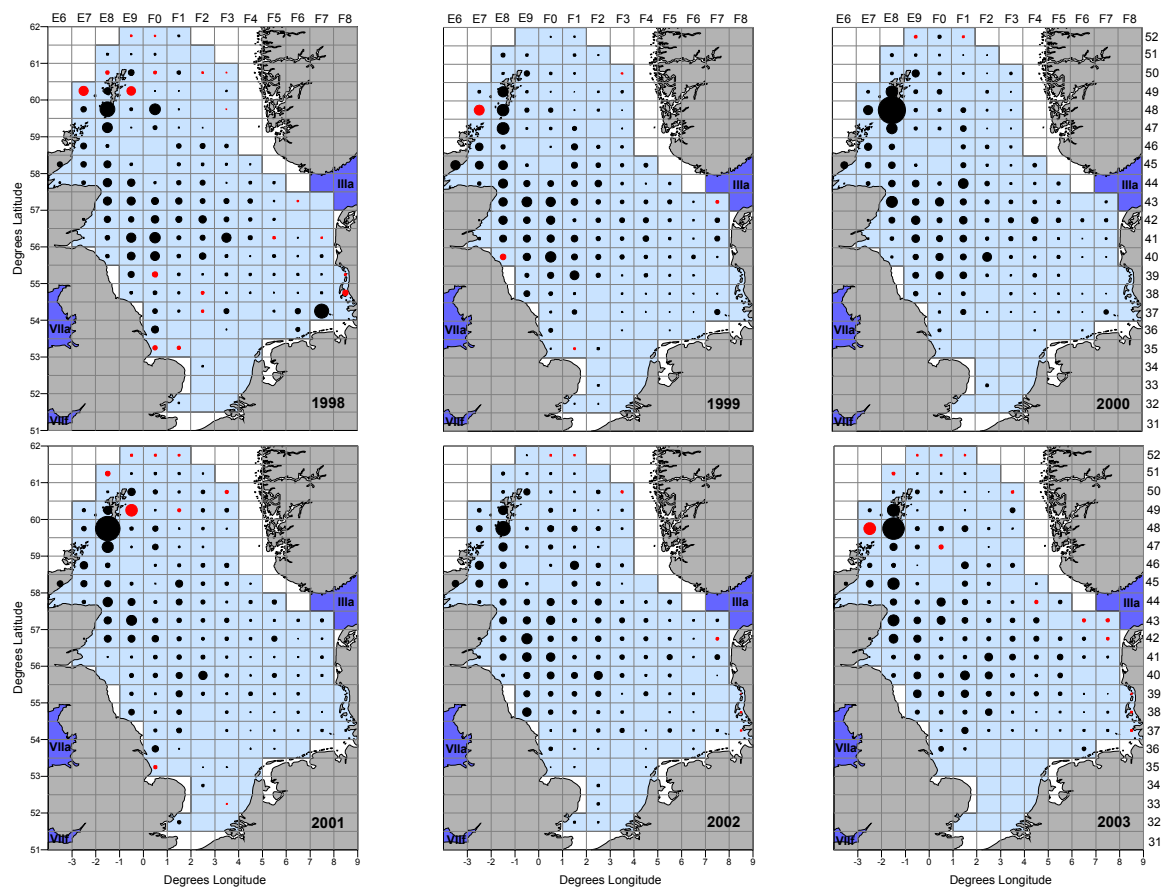


Figure 7.3.3.7. The distribution and estimated area-weighted catch of lemon sole in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 5,000 tonnes and is the same for all years.

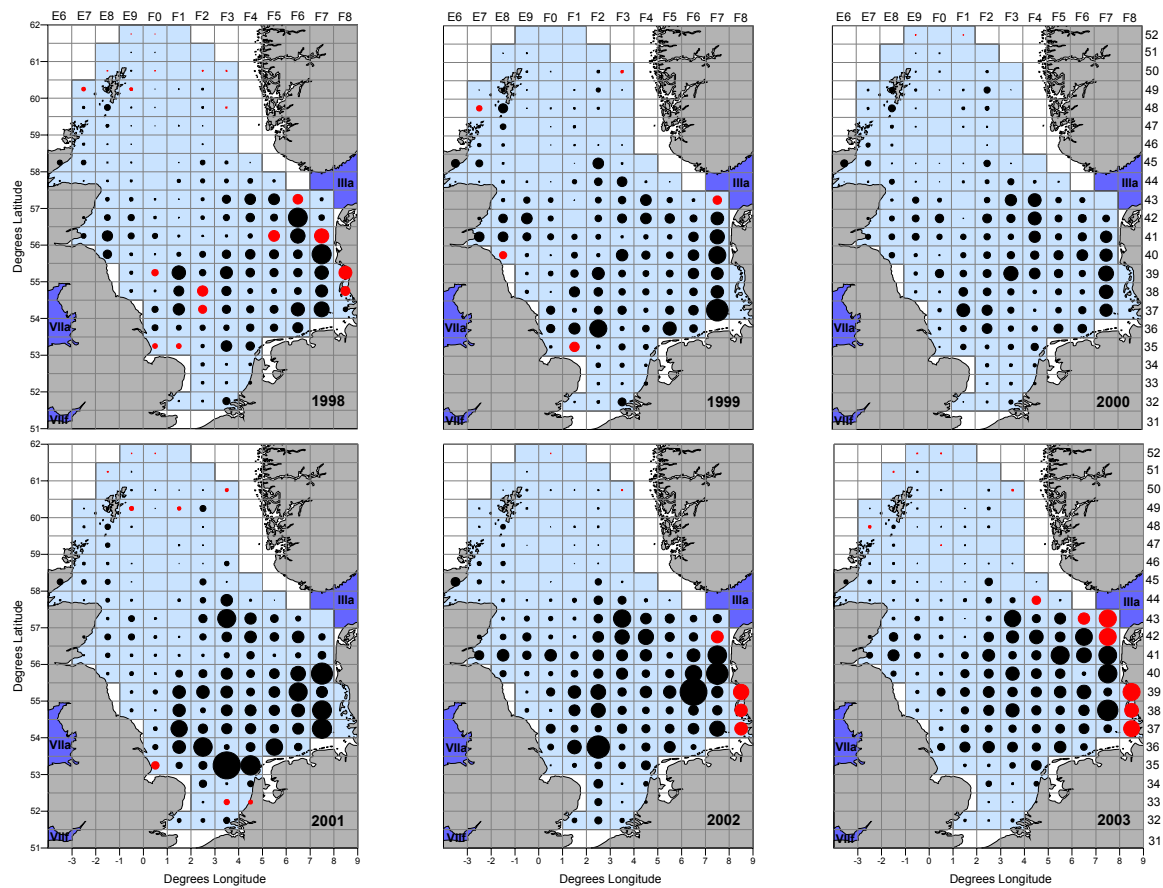


Figure 7.3.3.8. The distribution and estimated area-weighted catch of common dab in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 17,500 tonnes and is the same for all years.

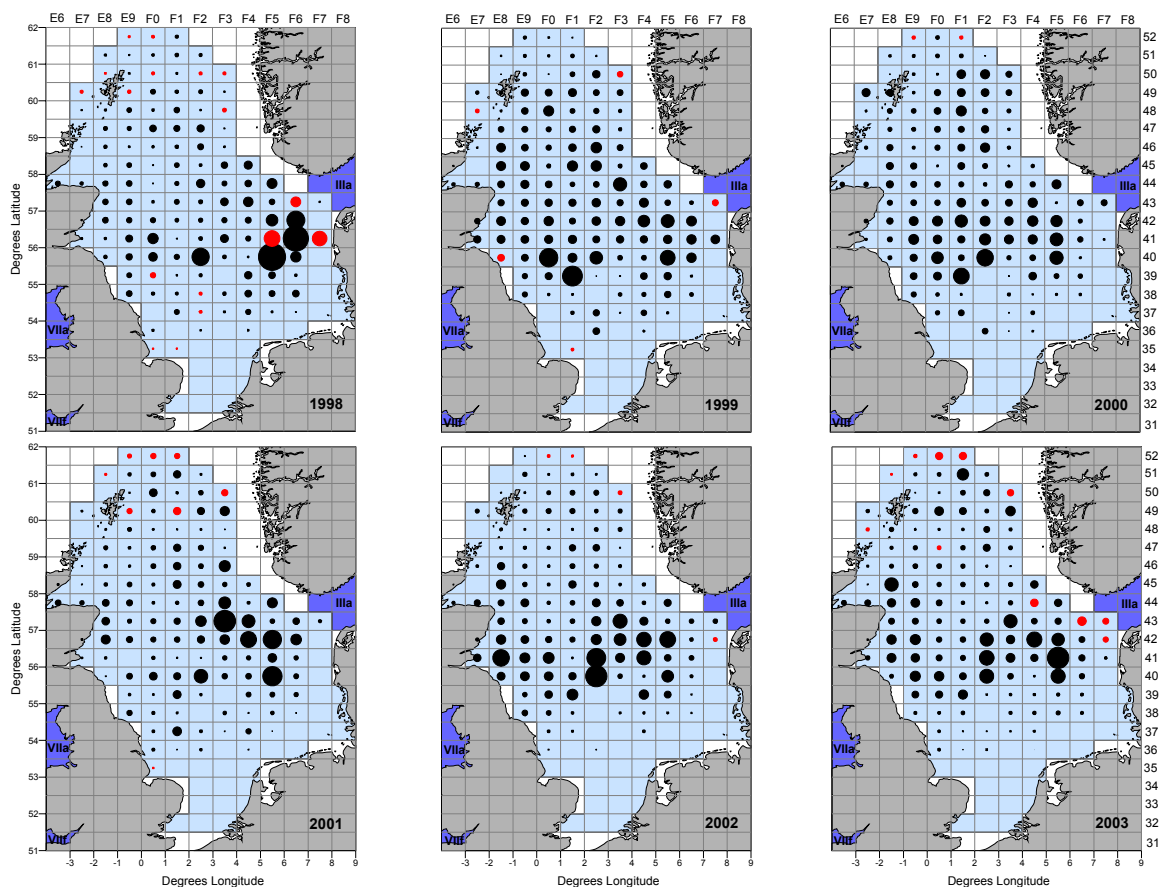


Figure 7.3.3.9. The distribution and estimated area-weighted catch of long rough dab in each statistical rectangle, 1998 – 2003 (Red symbols indicates interpolated data). Symbol size varies from 0 – 3,000 tonnes and is the same for all years.

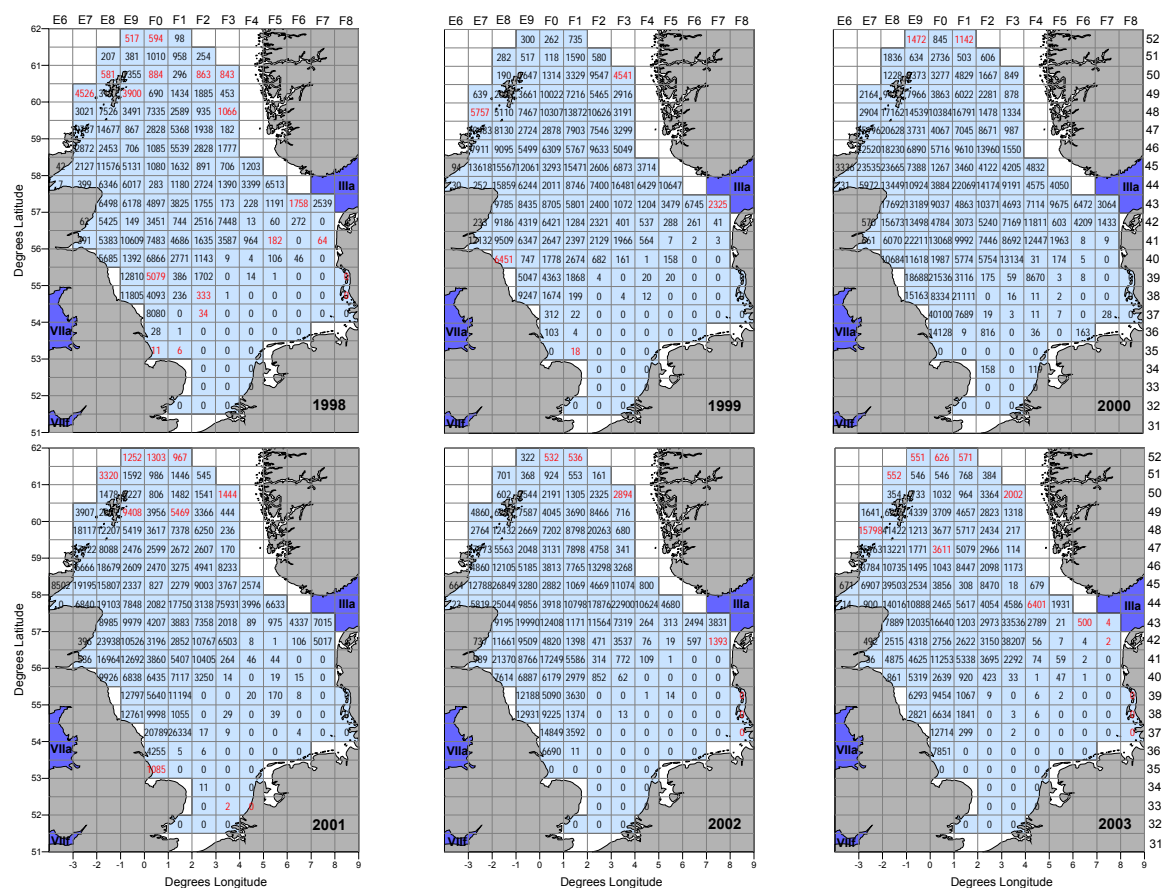


Figure 7.3.3.10. Estimated area-weighted catch (tonnes) of haddock in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

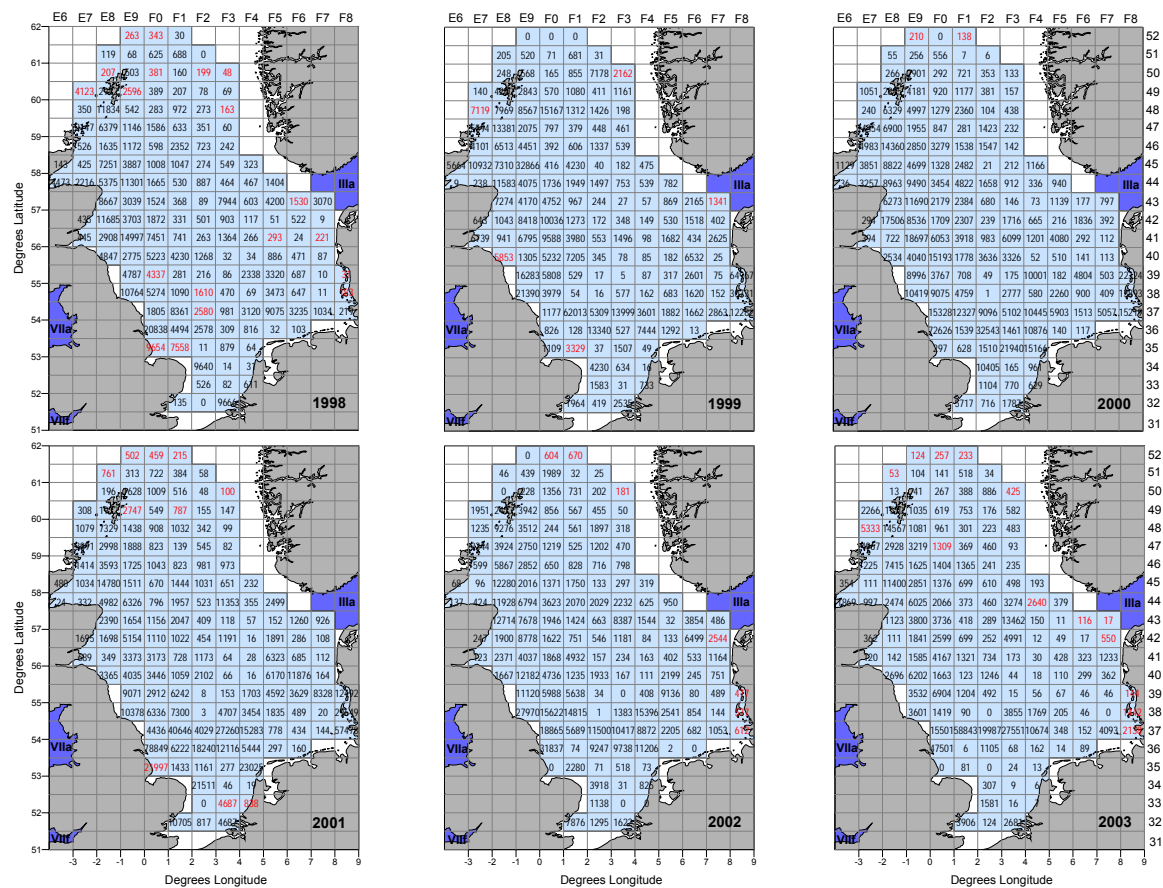


Figure 7.3.3.11. Estimated area-weighted catch (tonnes) of whiting in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

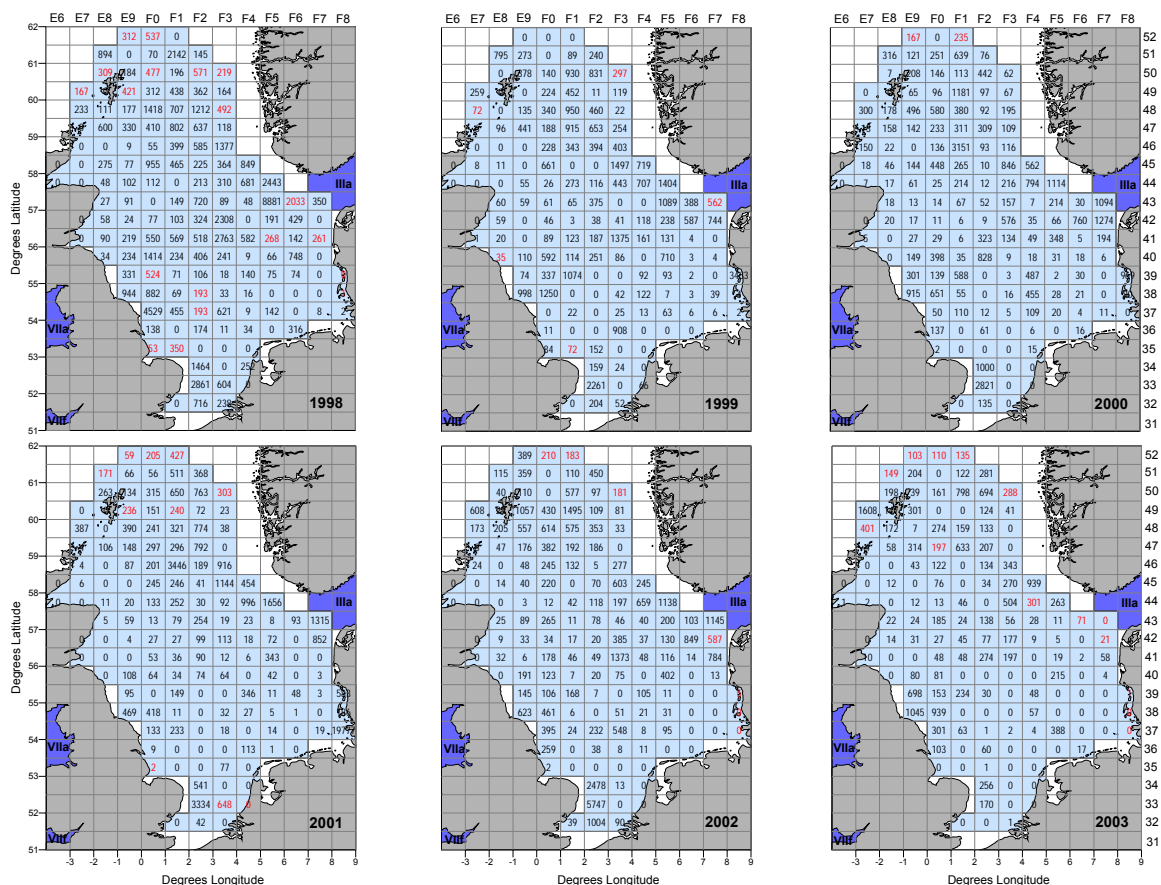


Figure 7.3.3.12. Estimated area-weighted catch (tonnes) of cod in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

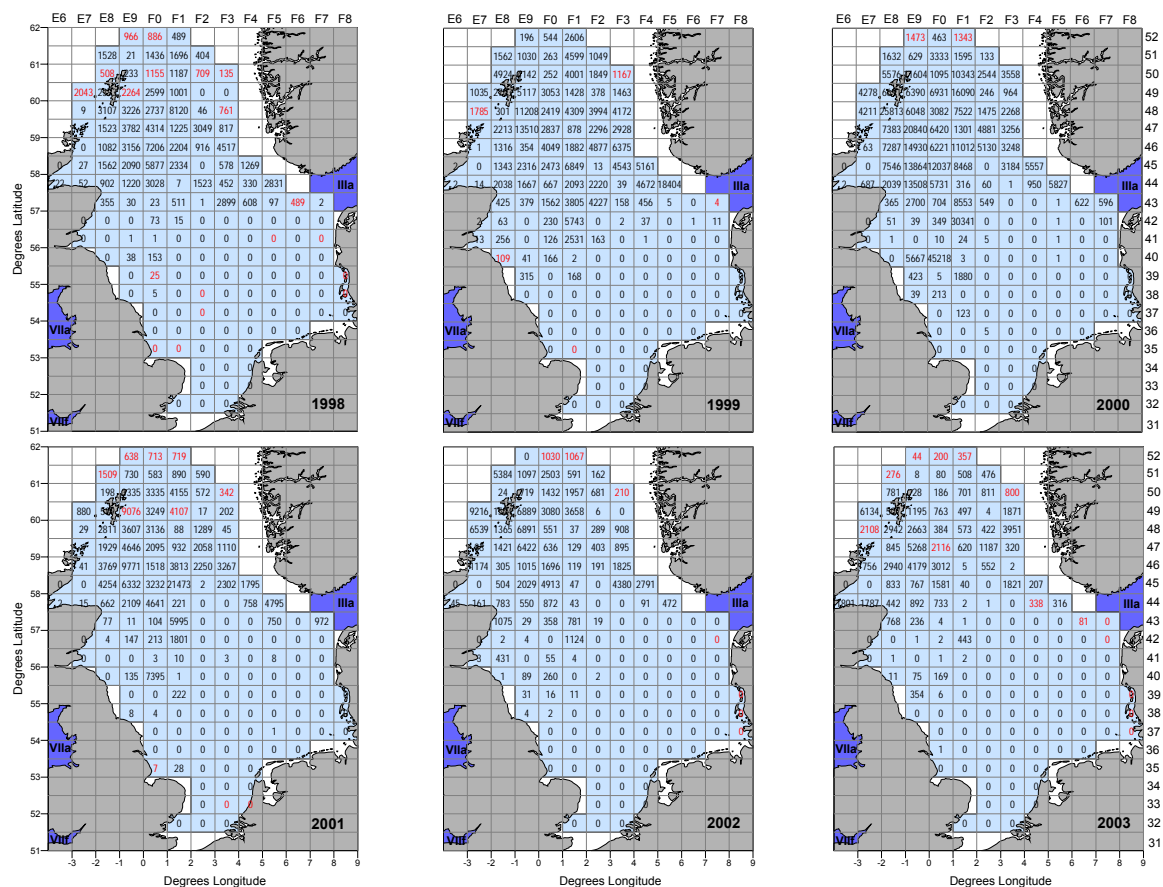


Figure 7.3.3.13. Estimated area-weighted catch (tonnes) of Norway pout in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

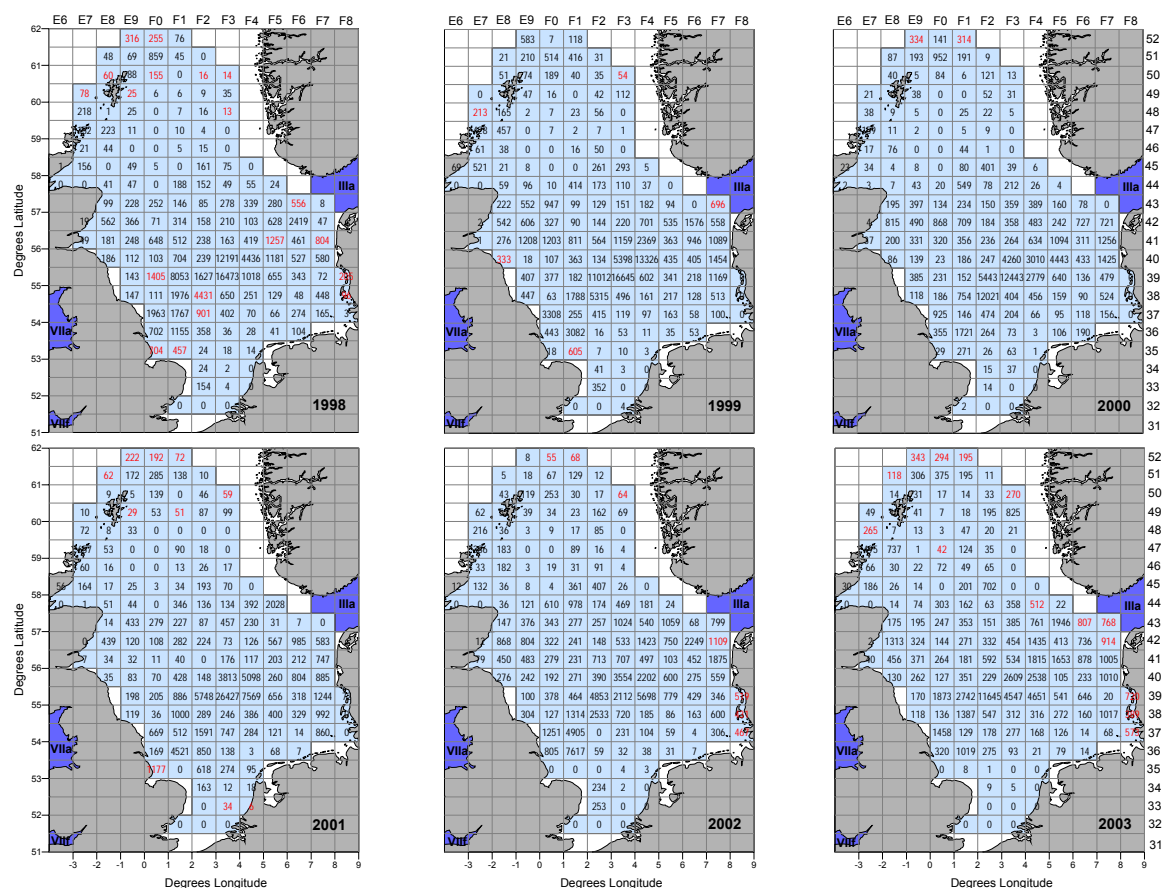


Figure 7.3.3.14. Estimated area-weighted catch (tonnes) of grey gurnard in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

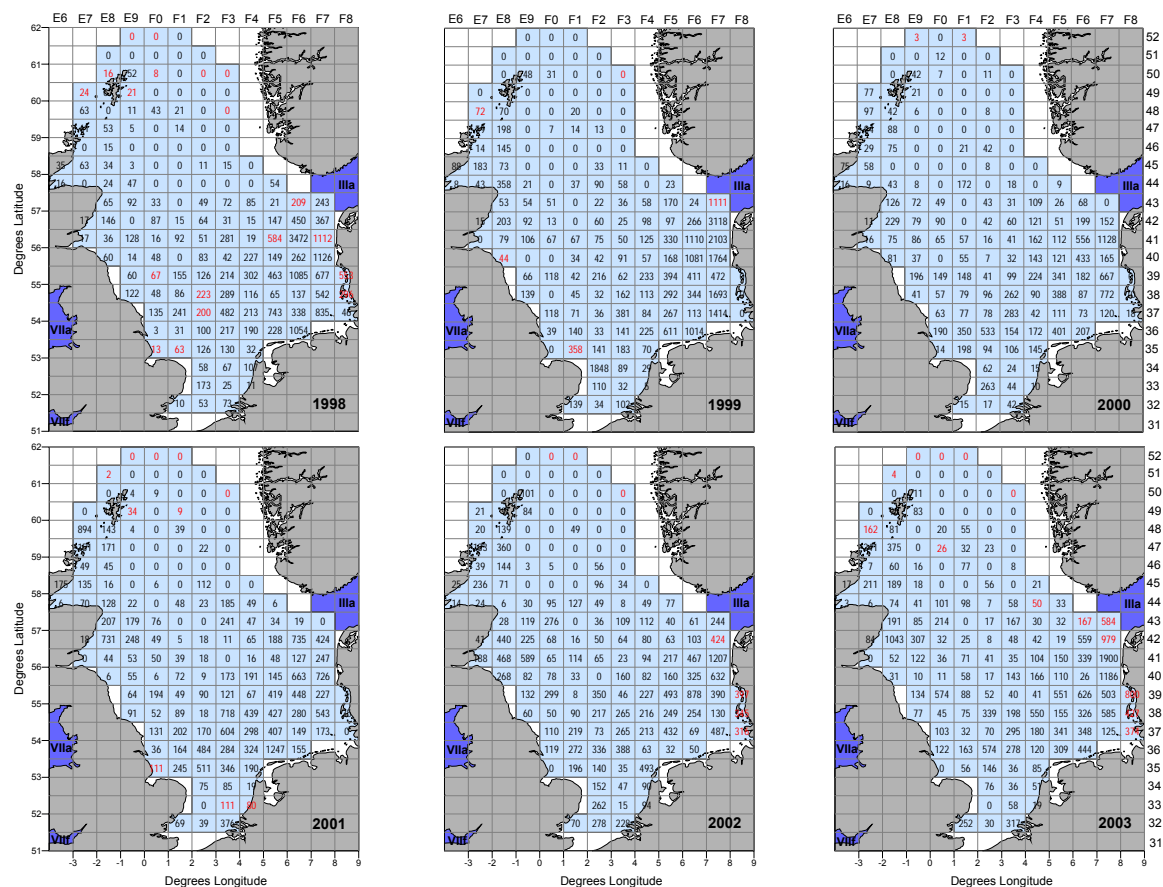


Figure 7.3.3.15. Estimated area-weighted catch (tonnes) of plaice in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

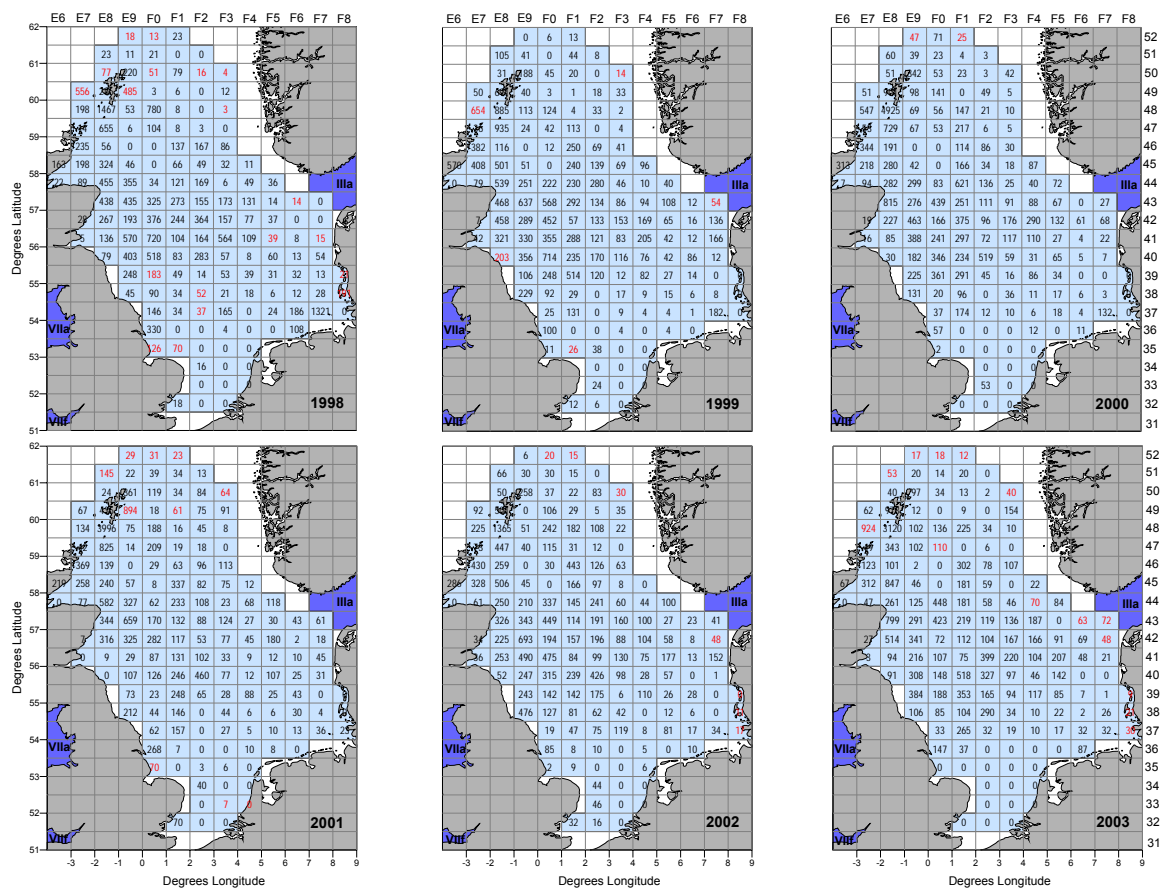


Figure 7.3.3.16. Estimated area-weighted catch (tonnes) of lemon sole in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

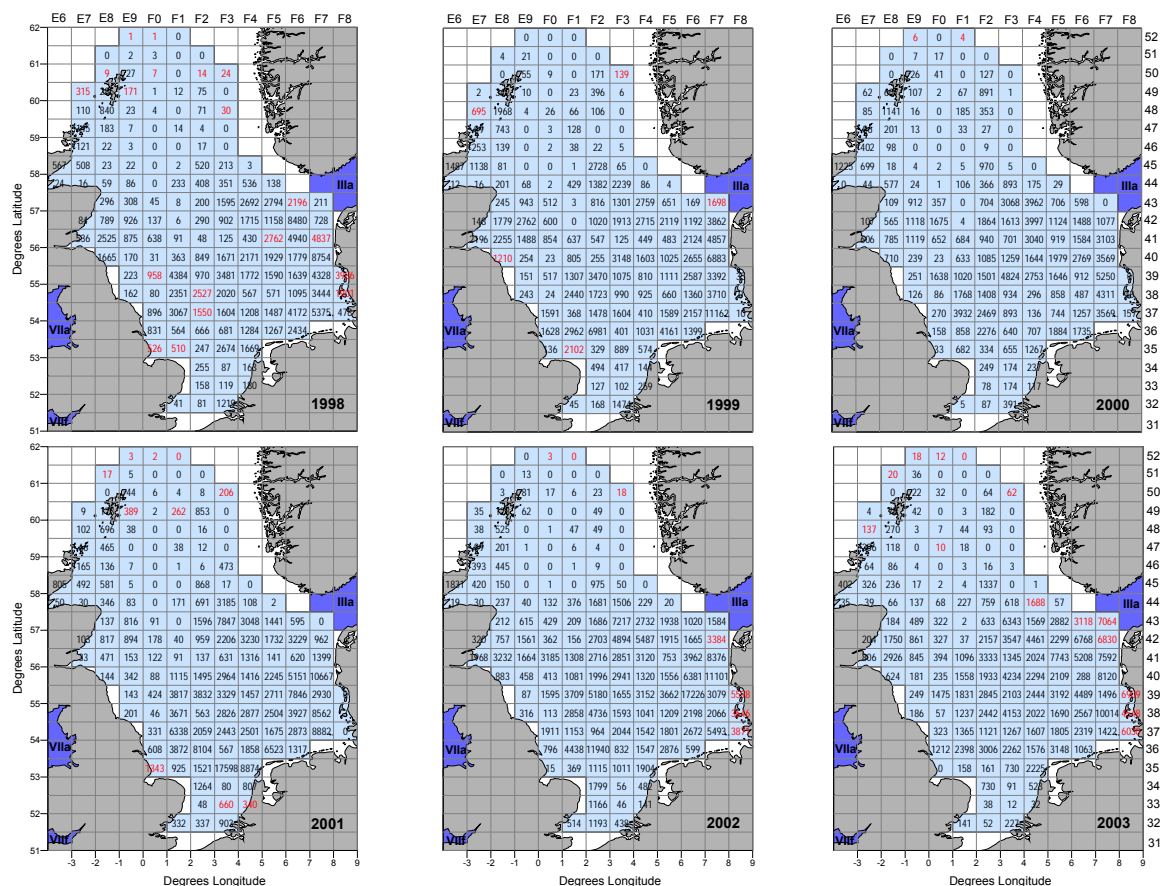


Figure 7.3.3.17. Estimated area-weighted catch (tonnes) of common dab in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

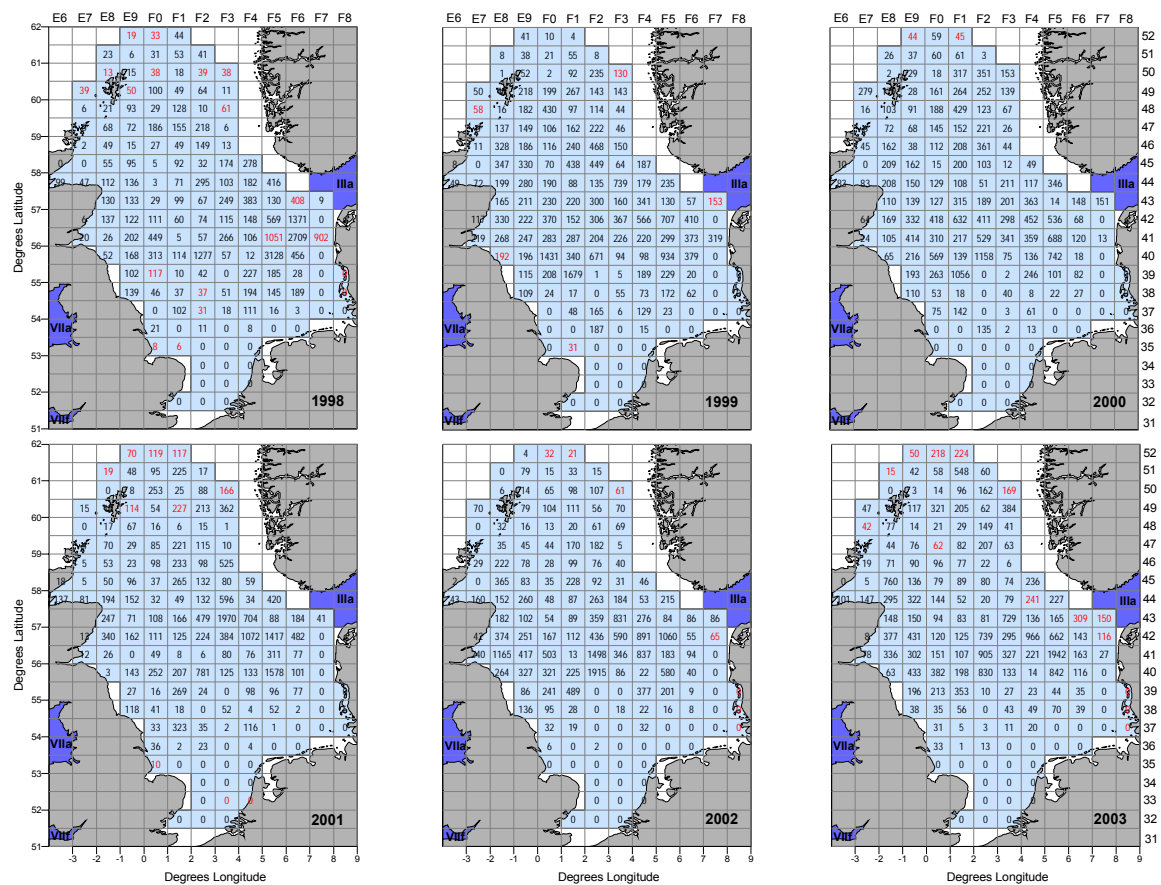


Figure 7.3.3.18. Estimated area-weighted catch (tonnes) of long rough dab in each statistical rectangle, 1998 – 2003. It should be noted that these are indices of relative abundance standardised to unit area, and are not estimates of total abundance.

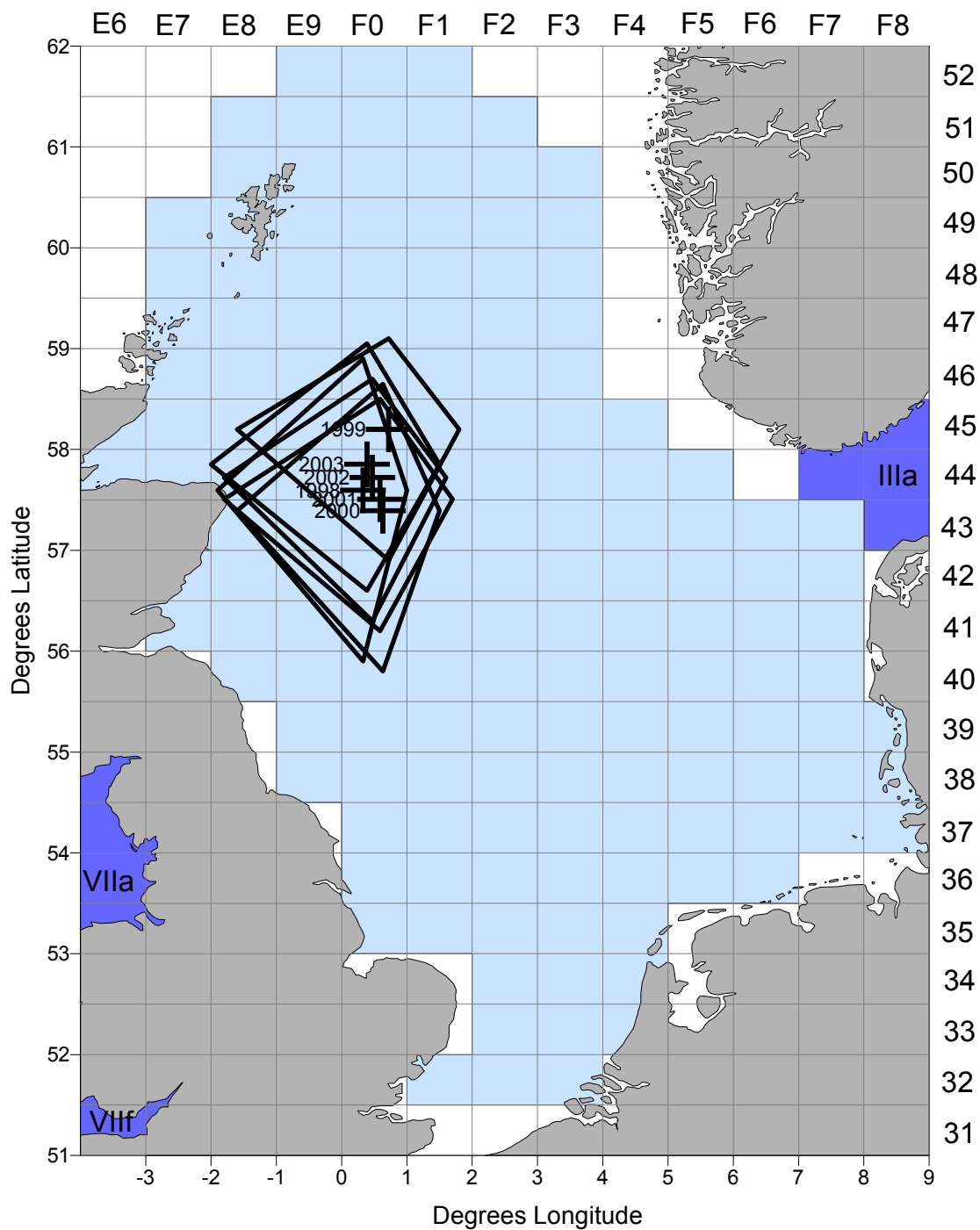


Figure 7.3.3.19. Geographical centroids for haddock in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

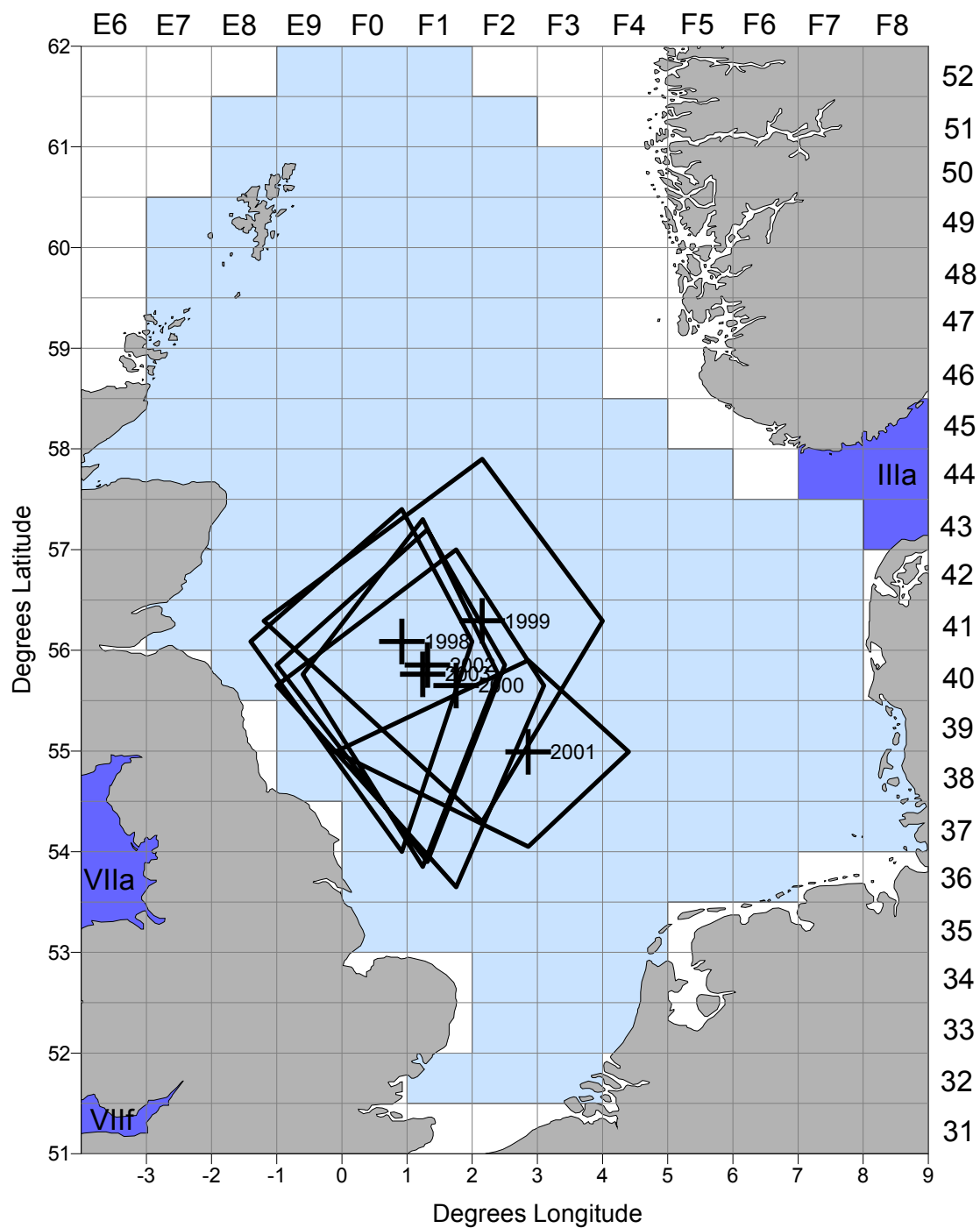


Figure 7.3.3.20. Geographical centroids for whiting in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

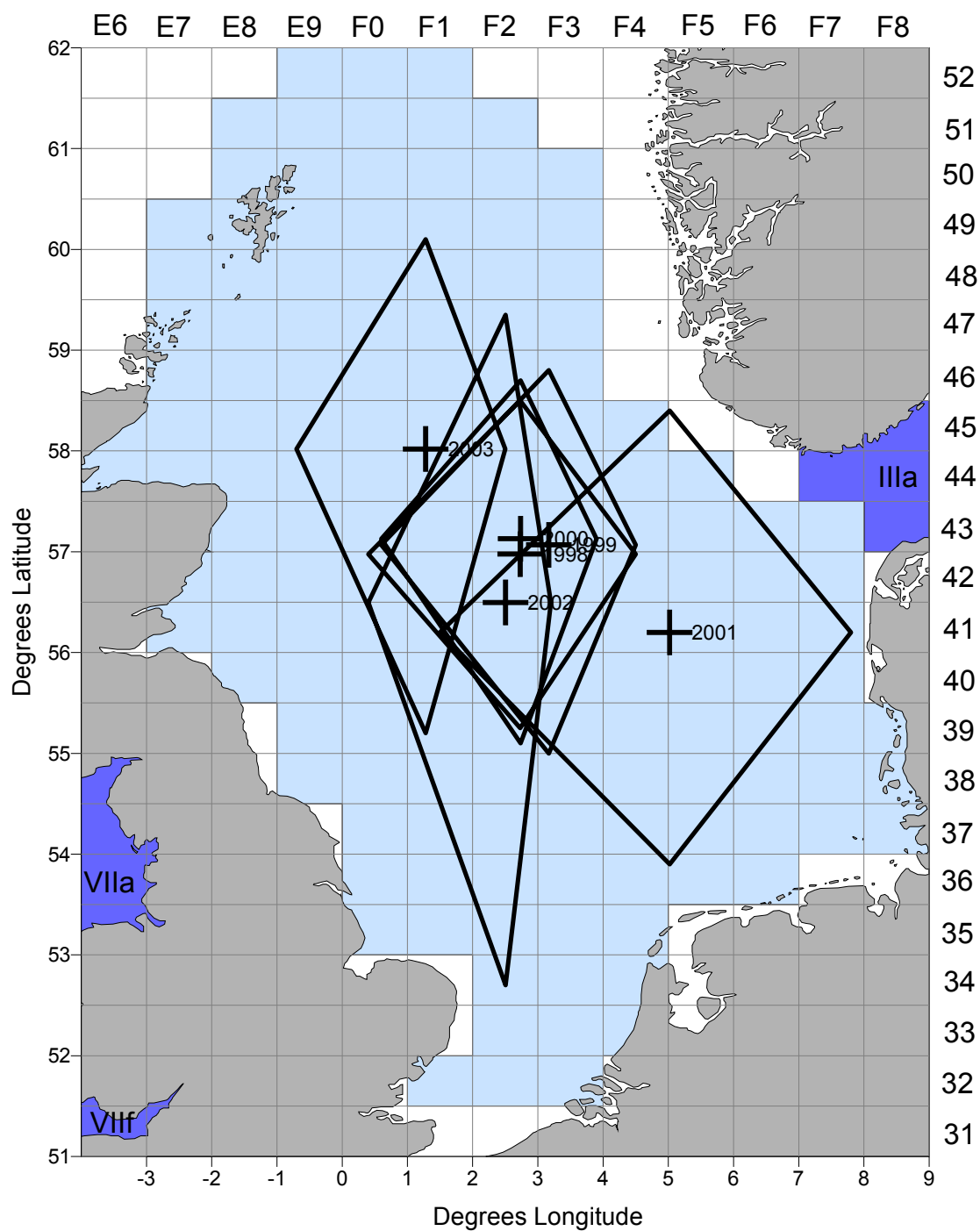


Figure 7.3.3.21. Geographical centroids for cod in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

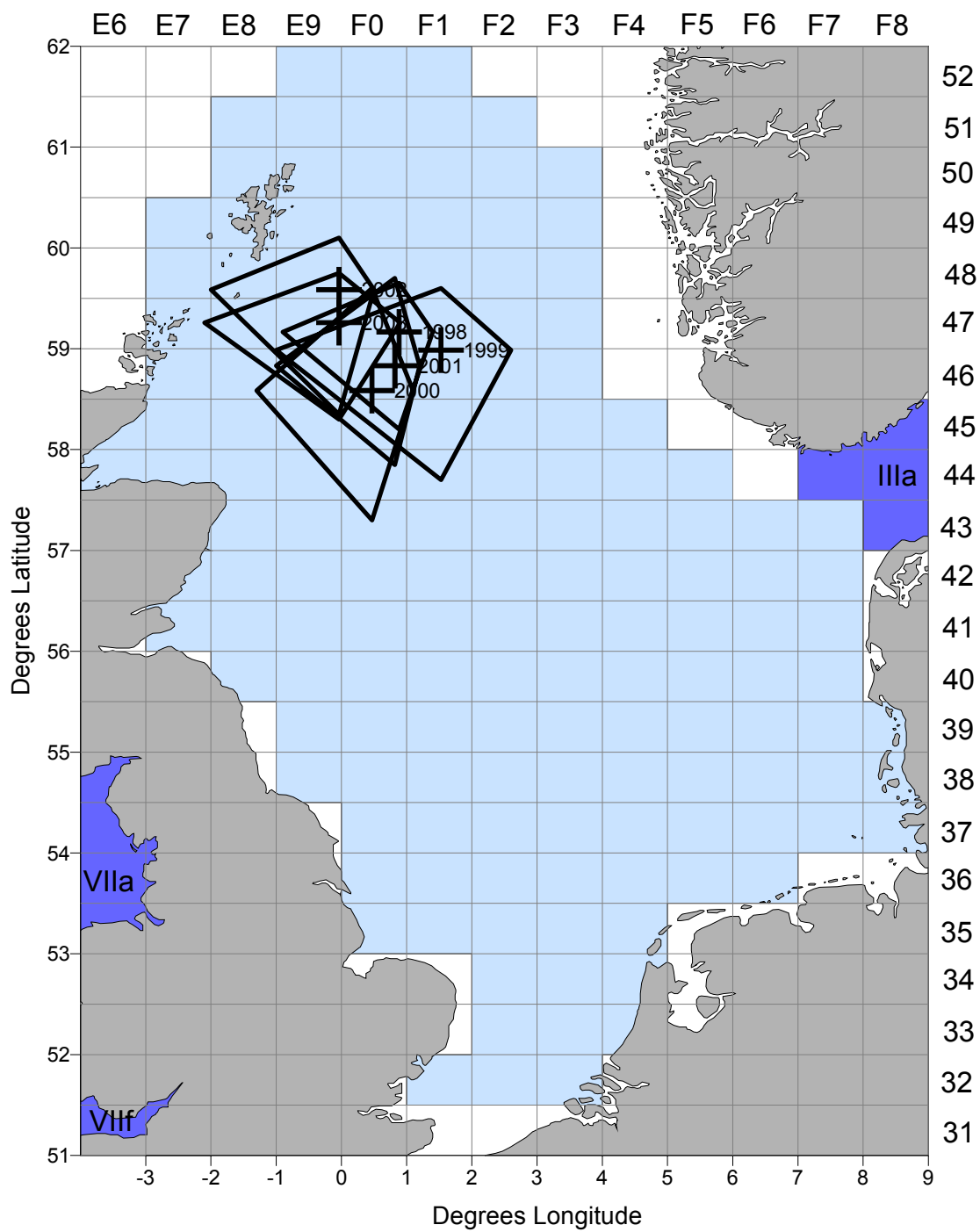


Figure 7.3.3.22. Geographical centroids for Norway pout in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

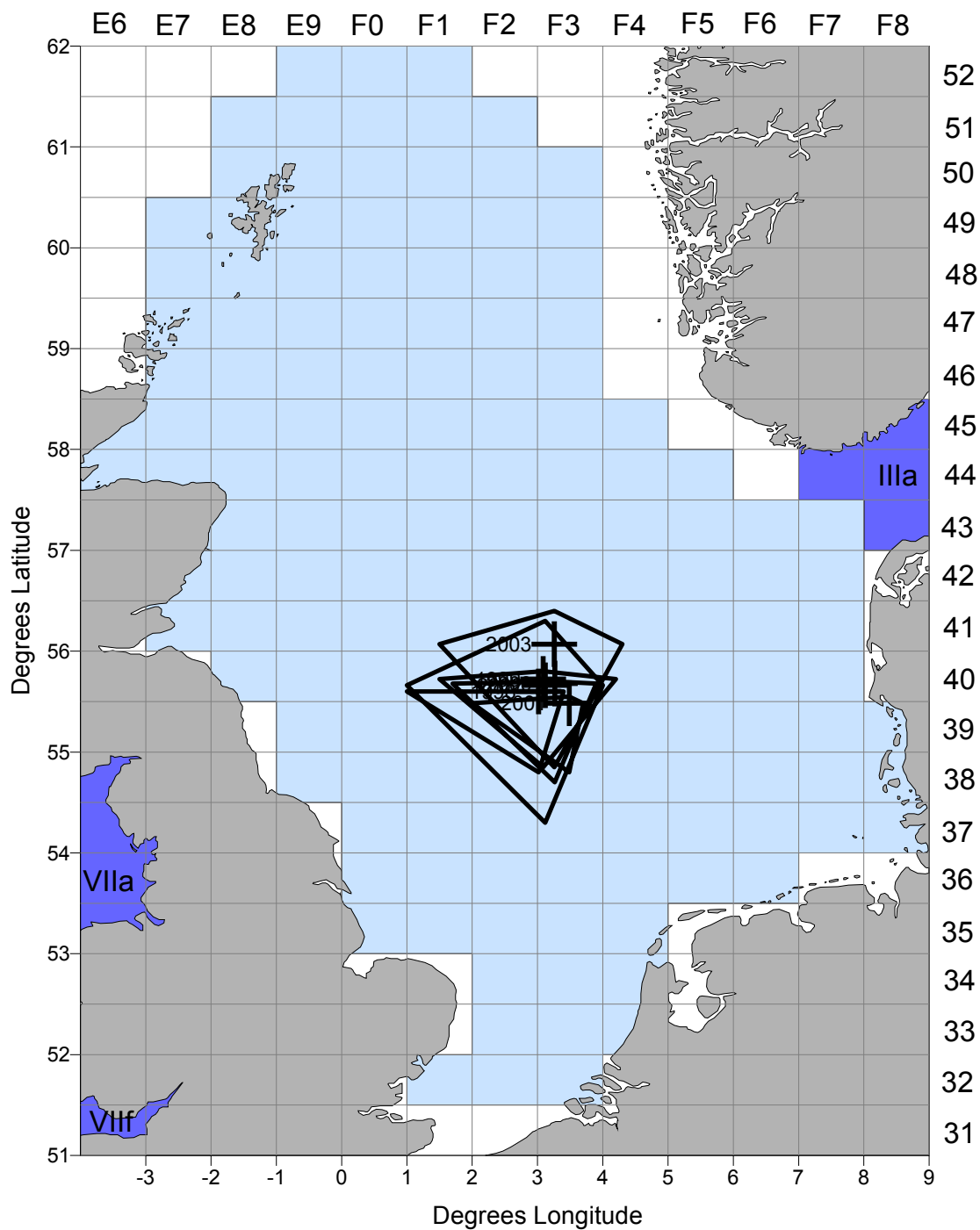


Figure 7.3.3.23. Geographical centroids for grey gurnard in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

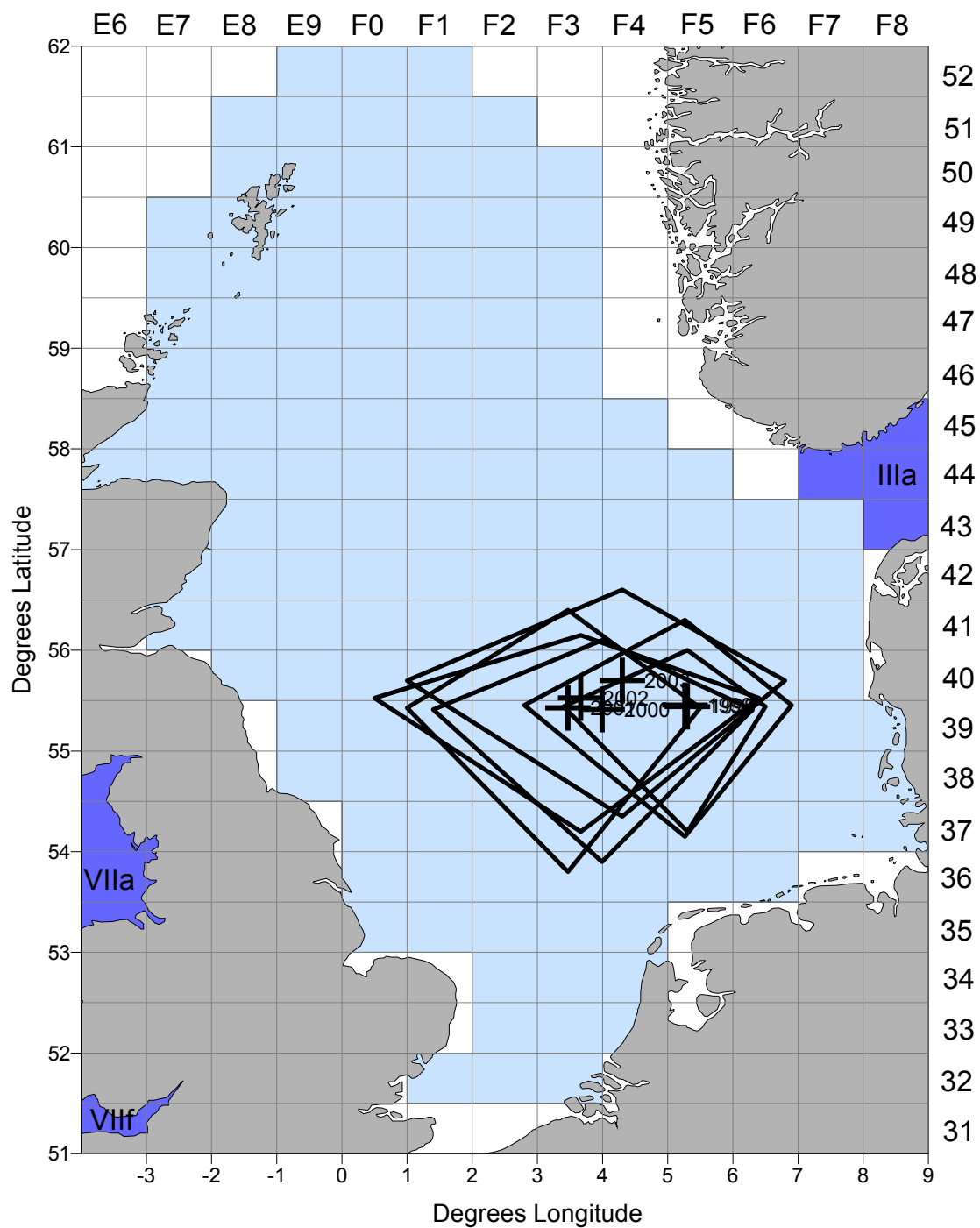


Figure 7.3.3.24. Geographical centroids for plaice in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

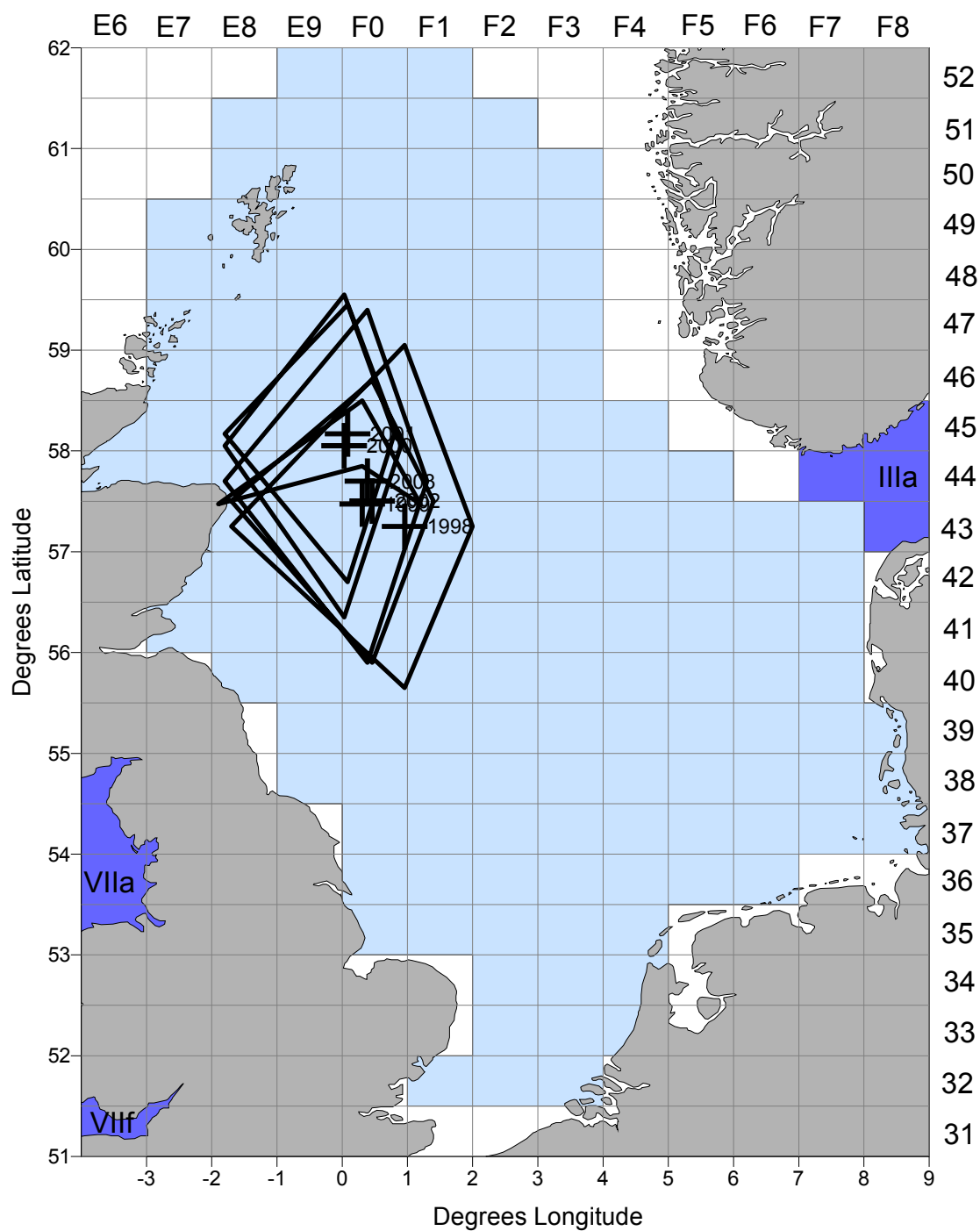


Figure 7.3.3.25. Geographical centroids for lemon sole in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

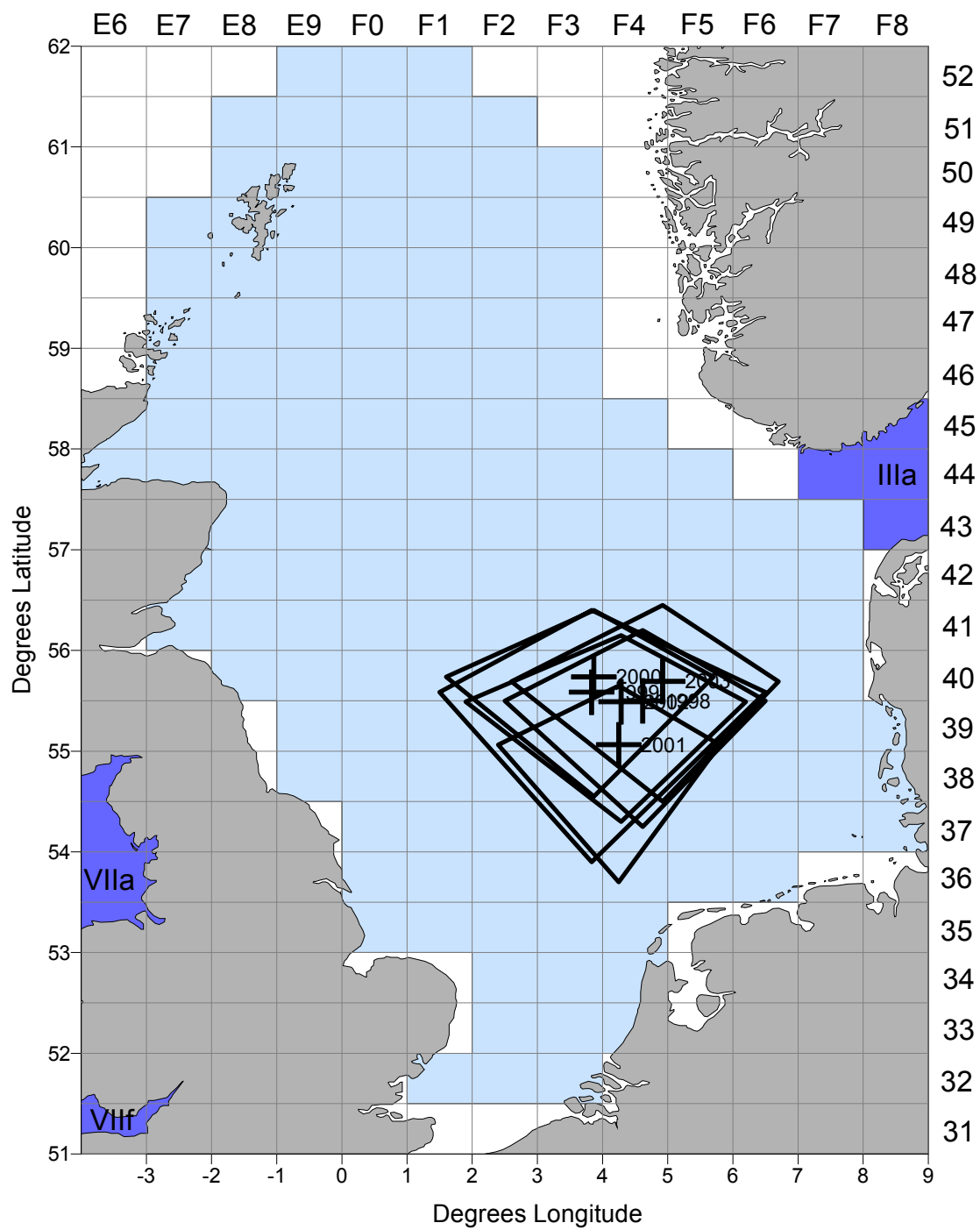


Figure 7.3.3.26. Geographical centroids for common dab in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

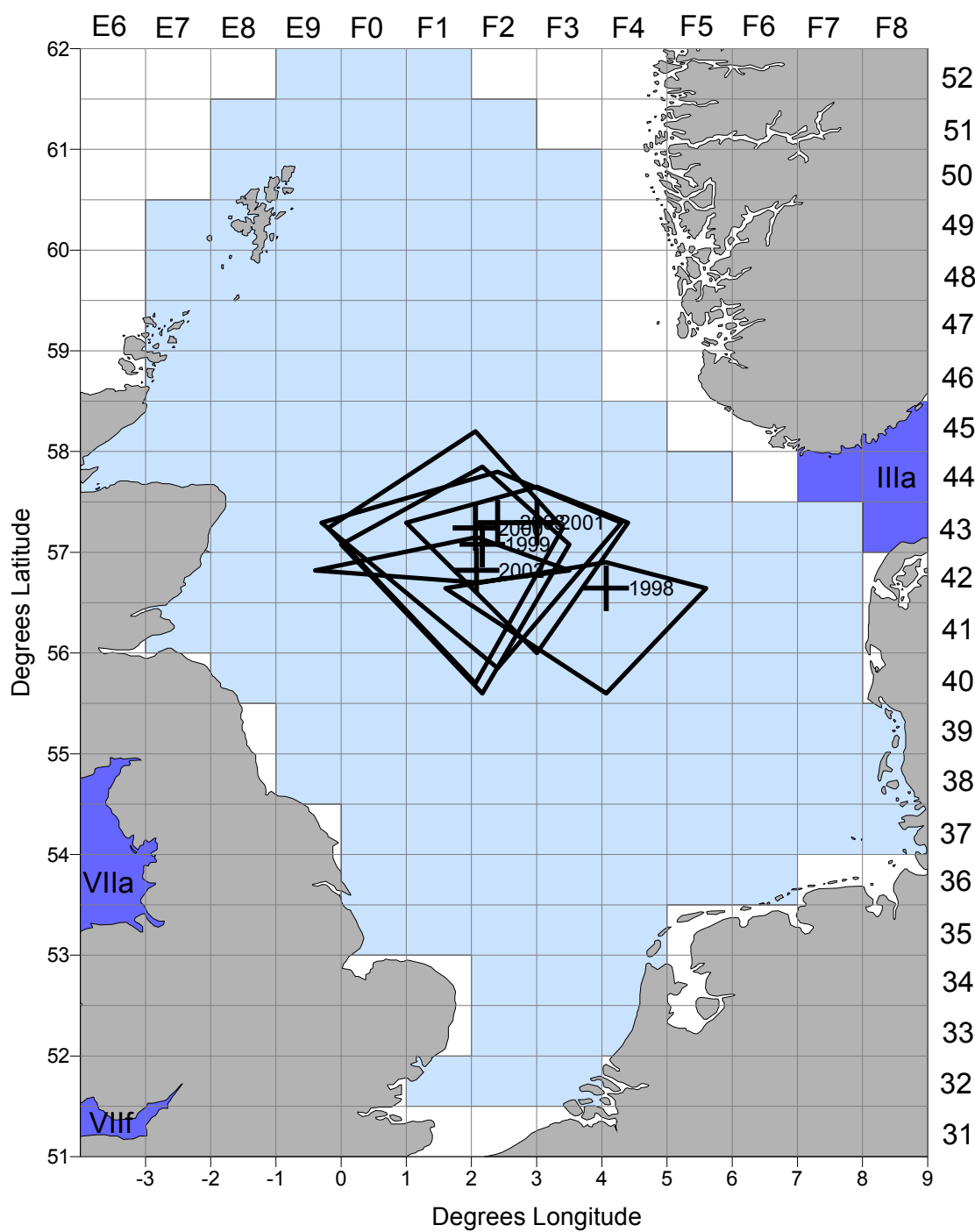


Figure 7.3.3.27. Geographical centroids for long rough dab in each year. These centroids give the mean central point location, along with latitudinal and longitudinal quartiles.

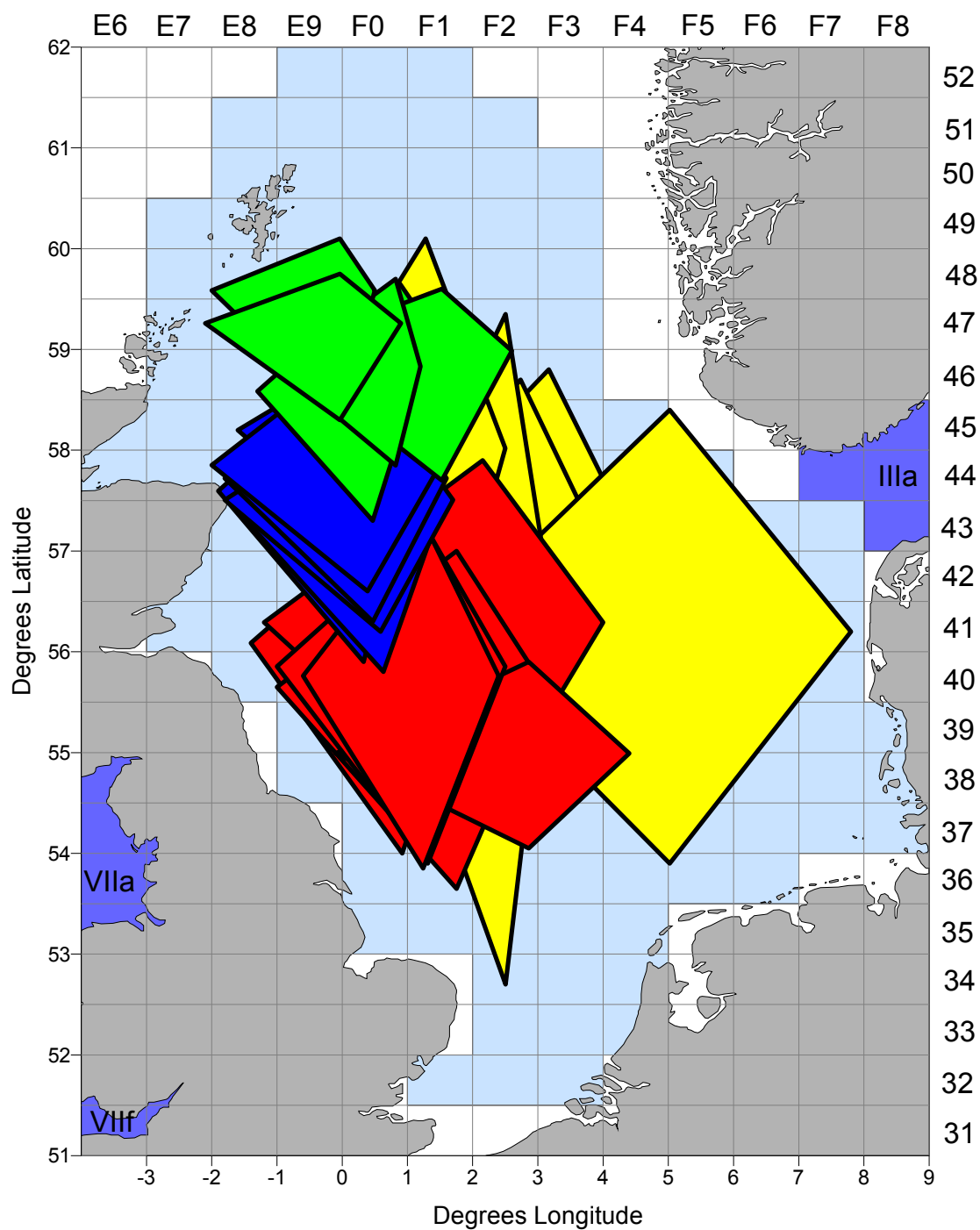


Figure 7.3.3.28. Geographical centroids for each of the gadoid fish in each year illustrating short-term variation in biogeographical distribution.

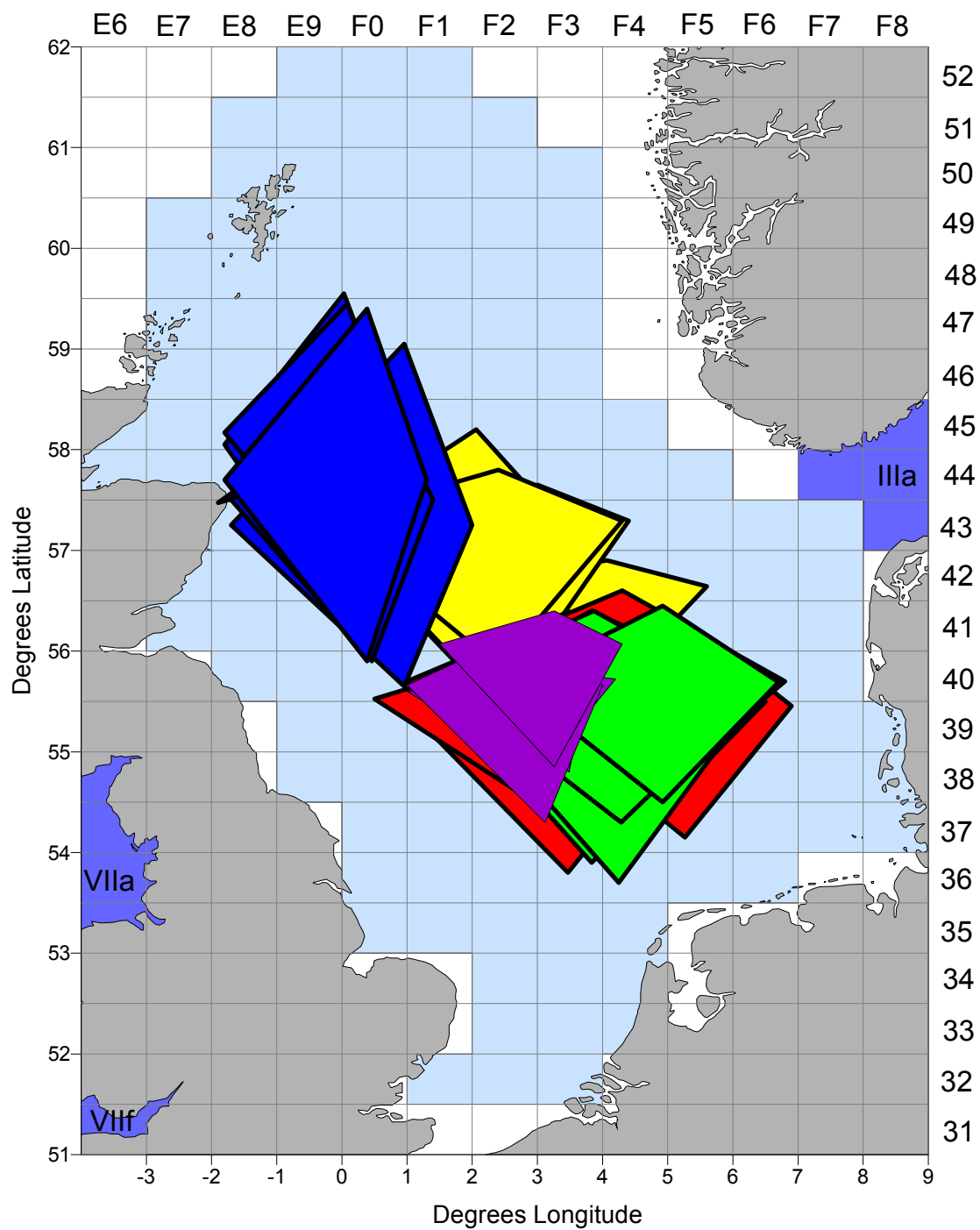


Figure 7.3.3.29. Geographical centroids for each of flat fish and grey gurnard in each year illustrating short-term variation in biogeographical distribution. variation in distribution.

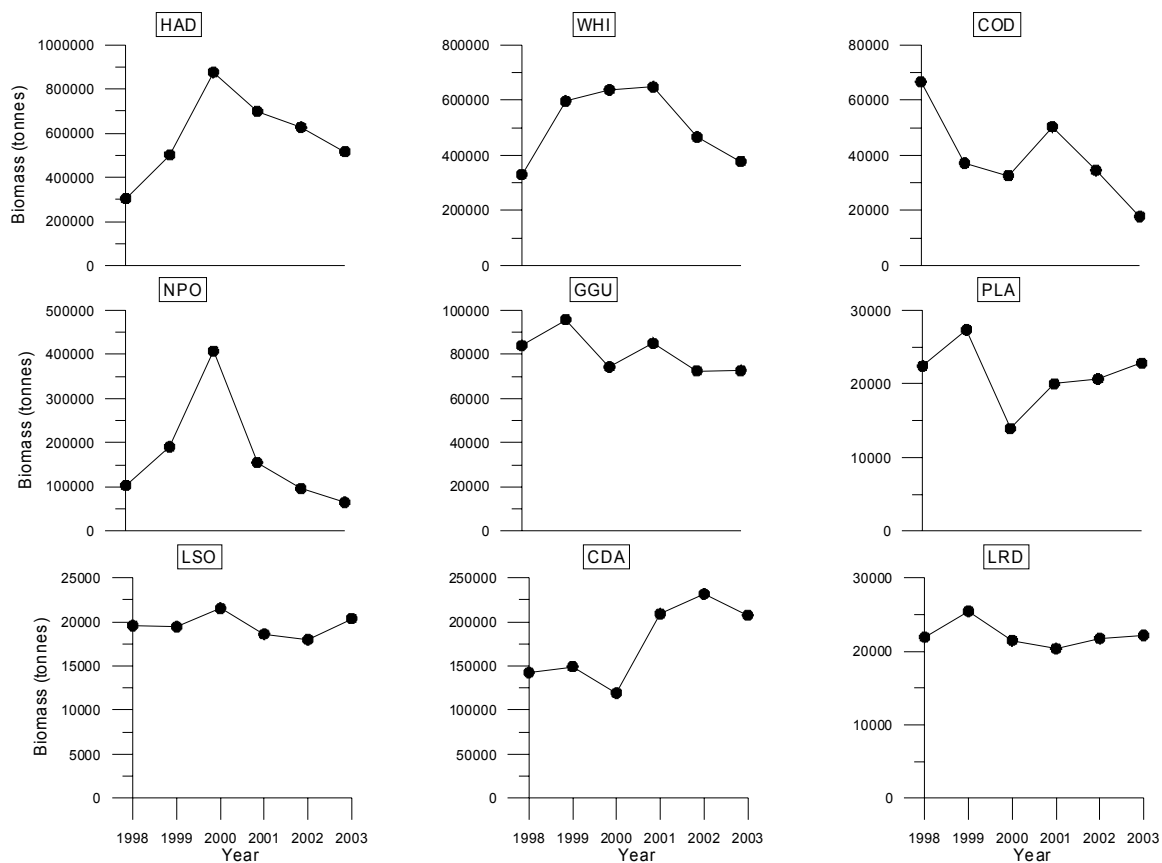


Figure 7.3.30. Estimated area-weighted catches for HAD (haddock), WHI (whiting), COD (Cod), NPO (Norway pout), GGU (grey gurnard), PLA (plaice), LSO (lemon sole), CDA (common dab), LRD (long rough dab), 1998 – 2003 using the IBTS 3rd Quarter data.

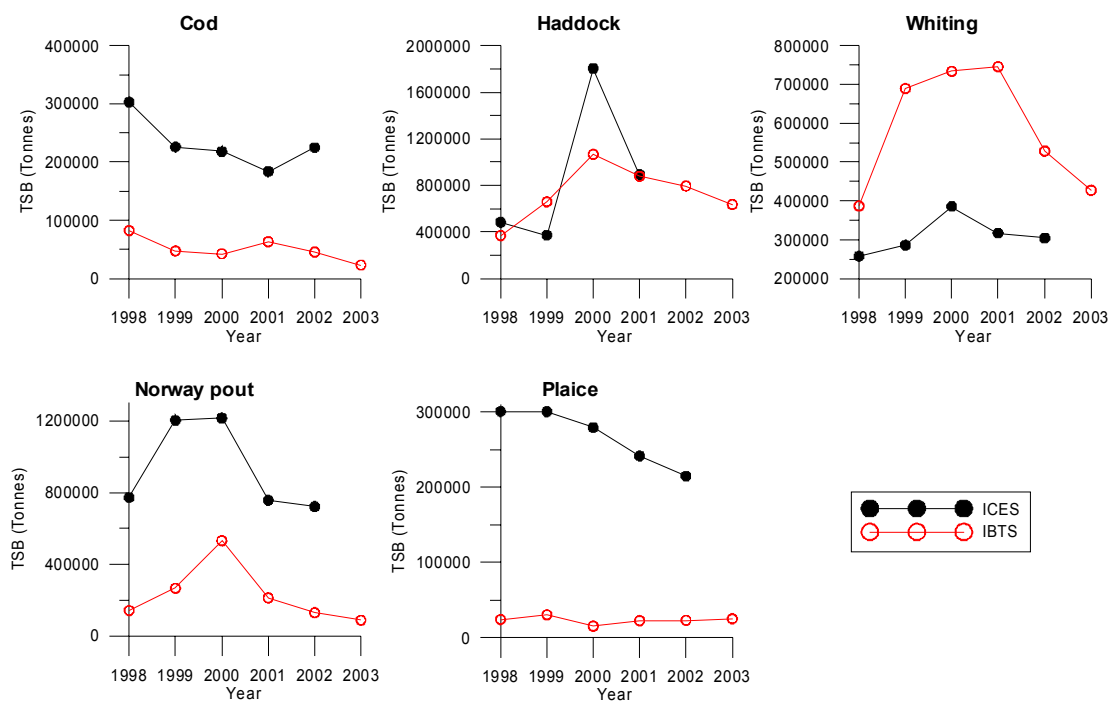


Figure 7.3.3.31. Comparison of IBTS-derived estimated of area-weighted catches, and biomass estimates used by ICES WGNSSK.

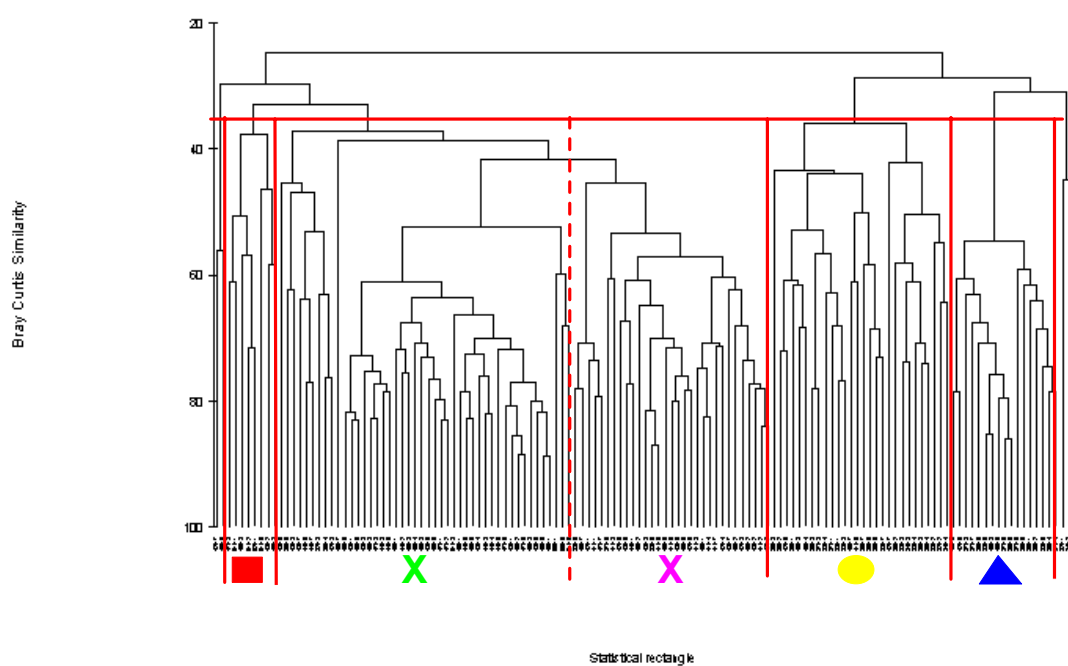


Figure 7.3.3.32. PRIMER dendrogram: clustering of statistical rectangles which have a similar species composition using the 1998 IBTS data. In this figure, four separate clusters are identified with a similarity of 35%. A sub cluster with a similarity of 40% is also indicated. The symbols correspond to those shown in Figures 7.3.3.33–34.

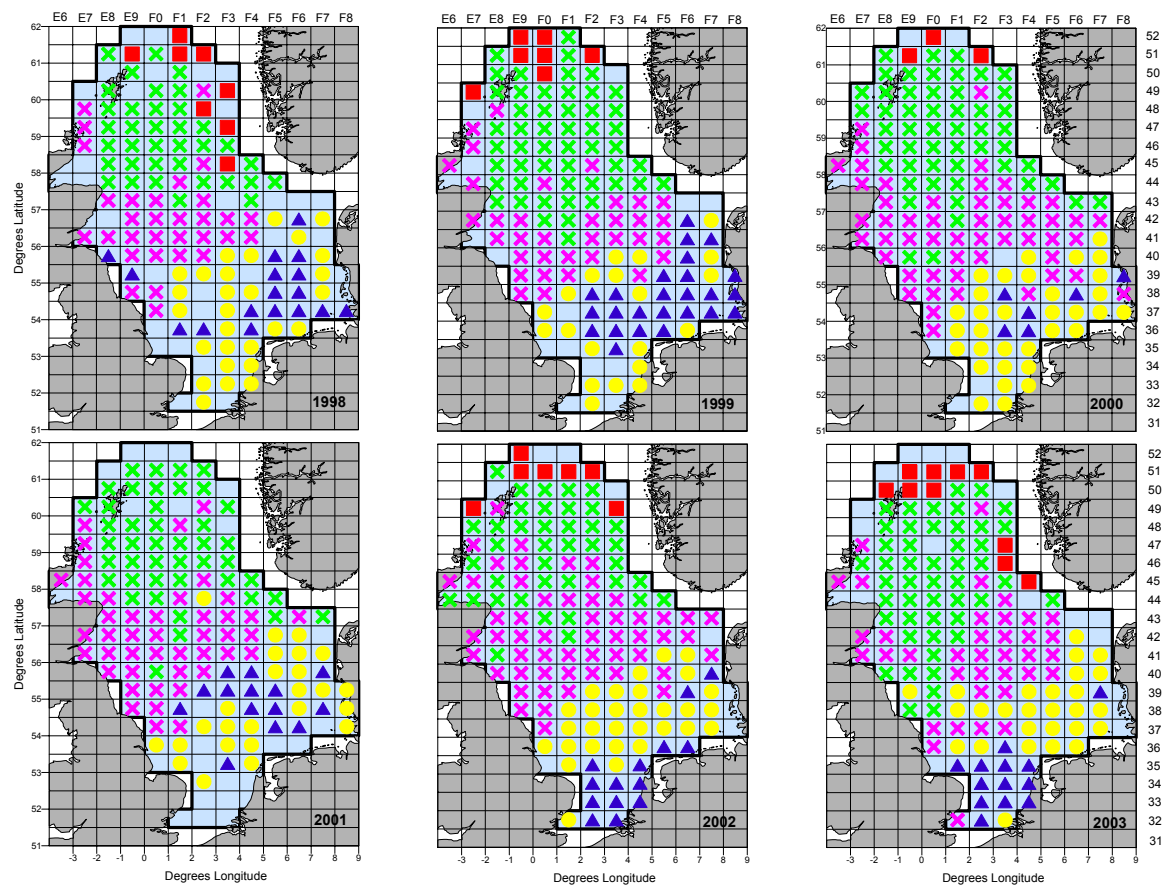


Figure 7.3.3.33. Results of the cluster analysis for each year 1998–2003. Clusters ■, ●, ▲ (X, X) have a similarity of 35%, X and X are sub-clusters at a similarity of 40%..

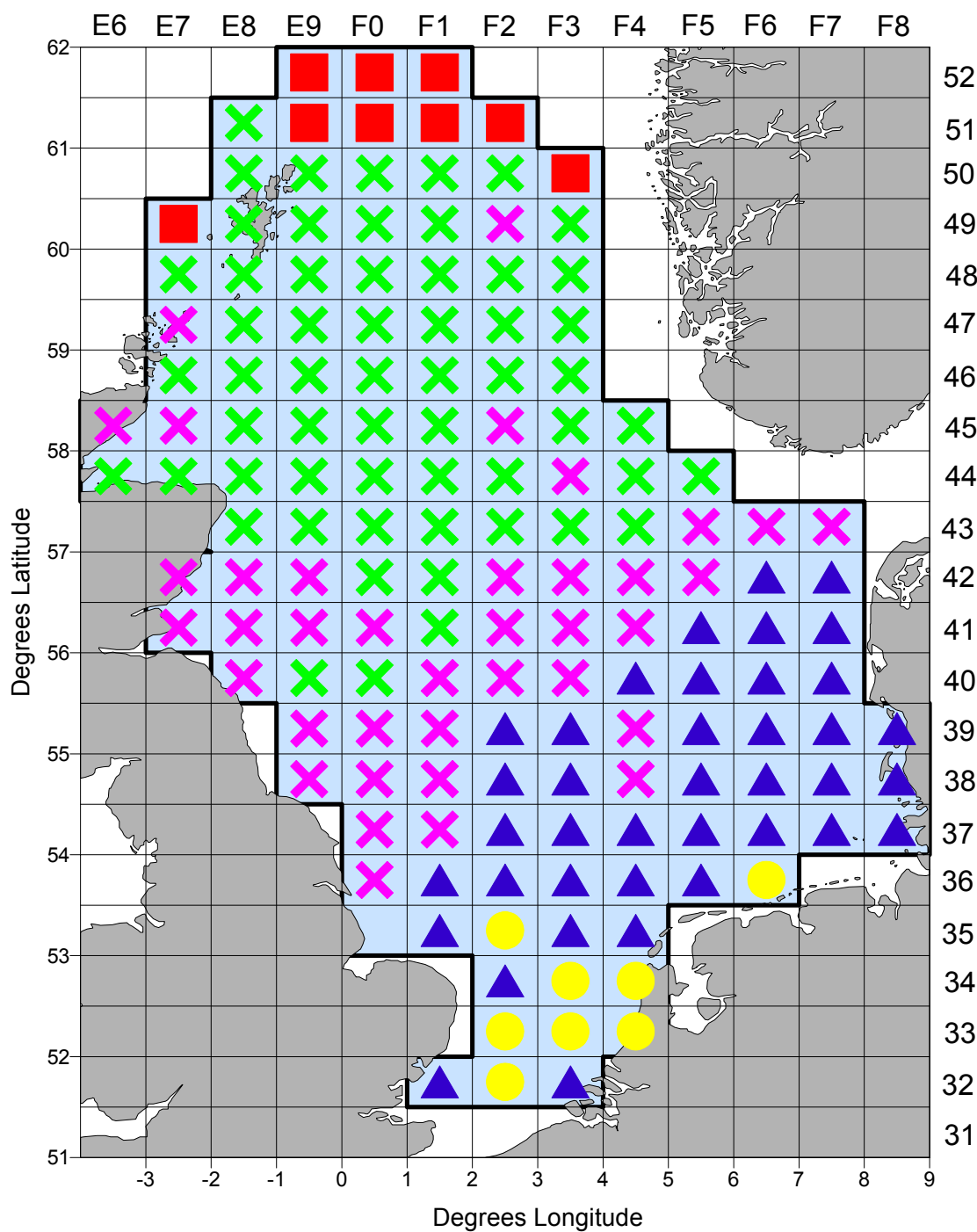


Figure 7.3.3.34. Results of the cluster analysis for all six years combined. Clusters ■, ●, ▲ (X, X) have a similarity of 35%, X and X are sub-clusters at a similarity of 40%.

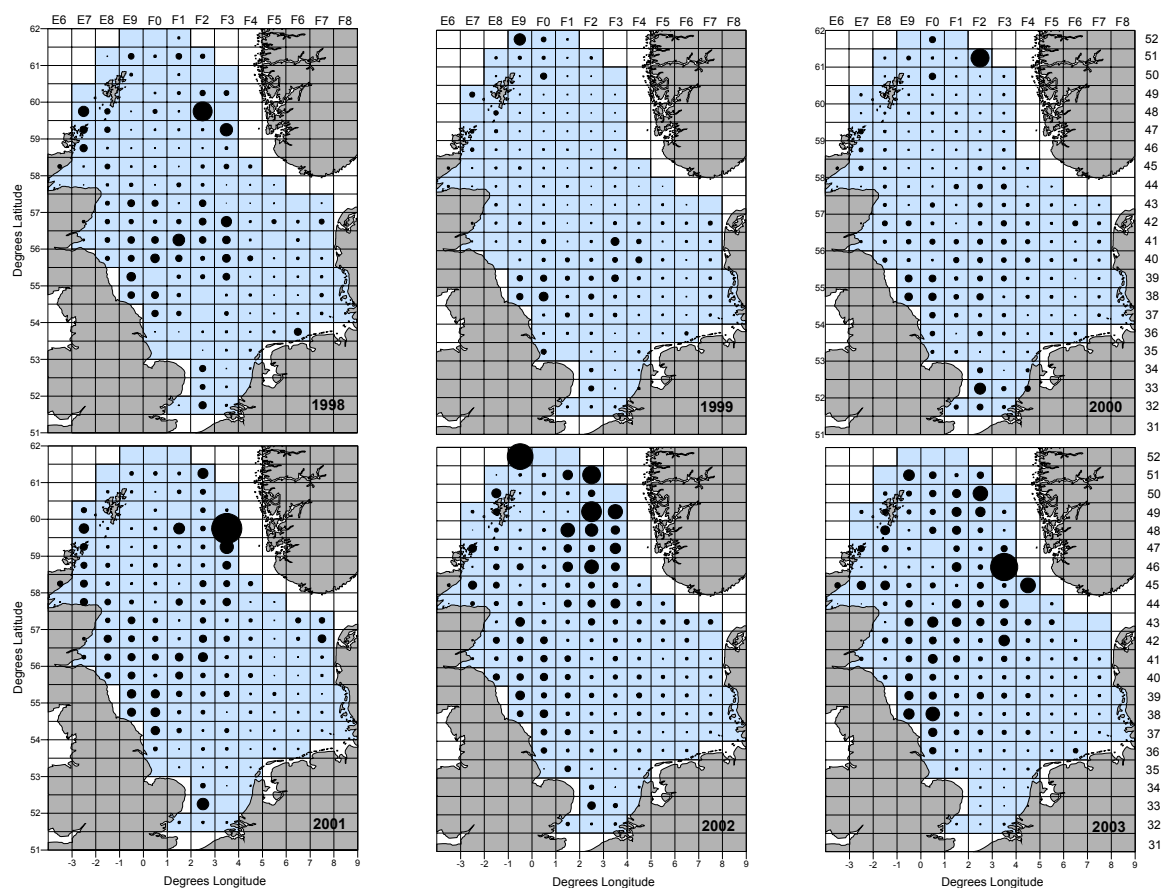


Figure 7.3.3.35. Spatial variation in mean weight (g) in each statistical rectangle from 1998 – 2003. Symbol size ranges from 0 – 627g.

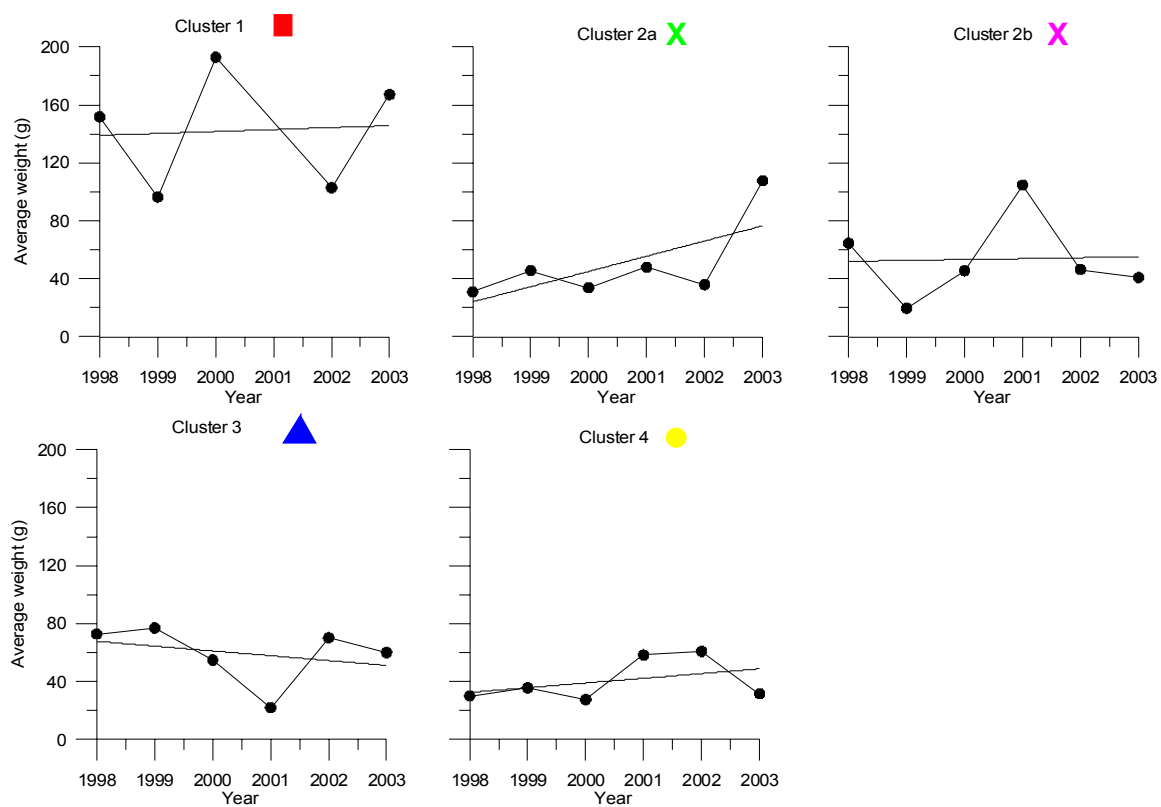


Figure 7.3.3.36. Inter-annual changes in mean weight (g) of demersal fish in each of the clusters.

8 CONCLUSIONS AND RECOMMENDATIONS

Issues regarding threatened fishes, fish communities and many other aspects of fish ecology are increasingly asked of ICES. Hence, it was considered that WGFE should meet next year. The following potential work areas for WGFE were suggested:

- A more holistic approach to EcoQOs could take the form of developing a suite of EcoQOs, each that is better targeted to a specific issue or concern, with these EcoQOs examined on a spatial scale appropriate to that issue. The development of EcoQOs is still required by OSPAR at the present time, and these need to be developed for fish communities and threatened and declining fish species. This work supports Action Point 2.2.2, and ultimately Action Points 2.2.1 and 3.2.
- Further work is required to explore abundance-range size relationships, including (a) the effect of sampling issues on the form of the abundance-occupancy relationship, (b) describing the intra- and interspecific abundance occupancy relationships of marine fishes, (c) the processes and mechanisms underlying marine abundance-occupancy patterns, and (d) exploring the utility of abundance-occupancy patterns in fisheries and ecosystem management issues. Abundance-range size relationships show clear potential links to other work covered by the group (e.g., fish habitat issues and the development of EcoQOs). This work supports Action Points 1.2.1 and 1.2.2.
- Review the updated IBTS manual and assess whether the quality assurance for IBTS data (e.g., for gear definition, taxonomic identification and survey protocols) is sufficiently robust to provide the data appropriate to examine ecosystem issues. Although the primary rationale for spatially comprehensive fishery-independent trawl surveys is to collect data for commercial fish species, studies on the overall fish, and benthic, communities is of increasing interest in ecosystem issues and management. This work supports Action Points 1.11 and 1.13.1, and will facilitate future studies pertaining to fish communities (e.g., 1.2 and 2.2).
- Continue studies on food rations and prey composition of North Sea fishes by (a) re-evaluating predation mortalities of the MSVPA prey fish populations, and examine the consequences by relevant runs of MSVPA/FOR when using food rations of MSVPA predators obtained by application of a new mechanistic gastric evacuation model rather than food rations used at present by the ICES, and (b) estimate food rations and prey compositions of grey gurnard, horse mackerel, and mackerel in the North Sea, applying new information about gastric evacuation rates. This work supports Action Points 1.2.1, 3.5 and 4.3.
- Review existing literature and available data sources for the diet, feeding habits and foraging behaviour of all fishes (target and non-target) in the North Sea and adjacent waters to evaluate available information on predator-prey interactions for ecosystem assessments. This work supports Action Points 1.2.1.
- Evaluate the decline criteria used by existing nature conservation organisations, and address any upcoming nature conservation issues for marine fishes. This work supports Action Points 1.2 and 2.2.
- Continue the descriptions of fish habitat, to support studies on threatened, commercial and selected non-target species. Such studies have implications to management issues and will also aid in the interpretation of abundance-range size relationships. This work supports Action Points 1.2.1 and 1.4.2.
- Continue studies on relative catchability of fishes (including the effects of fish size) in survey gears, and examine the implications of gear effect on the descriptions of the structure and function of fish assemblages. This work underpins many of the work areas above, therefore supporting studies in relation to Action Points 1.2, 1.11 and 1.13.
- The WG supports the approach by REGNS and wishes to continue liaison with the project, and to supply data and supporting text that is required for the North Sea and adjacent waters. This work supports Action Points 1.8
- Initiate studies on the broadscale temporal changes in fish communities along the European continental shelf of the eastern North Atlantic (e.g. the area covered by parts of ICES divisions VI-IX). Case studies presented in the present report have indicated that comparable signals, which may be related to environmental or oceanographic conditions, are operating in both the Celtic Sea and Portuguese waters. Hence, a more regional approach may provide a better understanding of the spatial-temporal dynamics of fish communities in these ICES areas. This work supports Action Points 1.2.2 and 1.6.

Annex 1: List of participants

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Annex 2 Working documents presented at WGFE

“Vulnerability of bottom fish species to the standard GOV” by Siegfried Ehrich (Institut für Seefischerei, Germany); Henning Reiss; Ulli Damm and Ingrid Kröncke

“Statistical power of the North Sea groundfish survey to detect trends in the abundance of rare and/or declining species” by David Maxwell and Simon Jennings (CEFAS, UK)

“Comparison and utility of different size-based metrics of fish communities for detecting fishery impacts” by Daniel Duplisea (DFO, Canada) and Martin Castonguay

“Relative catch efficiencies of selected fish species in survey gears” by Niels Daan (RIVO, The Netherlands)

“Indicators using L_{\max} for Portuguese continental waters” by Louize Hill and Maria de Fátima Borges